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WATER QUALITY CRITERIA 1972

National Academy of Sciences-National
Academy of Engineering

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National Academy of Sciences-National Academy of Engineering, Washington, D. C., Environmental Studies Board.

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--In 1971, at the request of the Environmental Protection Agency, the National Academy of Sciences-National Academy of Engineering undertook the revision of WATER QUALITY CRITERIA, the 1968 Report of the National Technical Advisory Committee (NTAC) to the Secretary of the Interior. The Academies appointed a Committee on Water Quality Criteria and six Panels, and the responsibility for overseeing their activities was assigned to the Environmental Studies Board, a joint body of the Academies. The guidelines for the Academies' Committee were similar to those followed by the NTAC. The Committee's six Panels were: (1) Recreation and Aesthetics, (2) Public Water Supplies, (3) Freshwater Aquatic Life and Wildlife, (4) Marine Aquatic Life and Wildlife, (5) Agricultural Uses of Water, and (6) Industrial Water Supplies. In the 1972 Report many new subjects are discussed in detail, among them: the recreational impact of boating, levels of use, disease vectors, nuisance organisms, and aquatic vascular plants; viruses in relation to public water supplies; effects of total dissolved gases on aquatic life; guidelines for toxicological research on pesticides and uses of toxicants in fisheries management; disposal of solid wastes in the ocean; use of waste water for irrigation; and industrial water treatment processes and resultant wastes. Many toxic or potentially toxic substances not considered by the NTAC are discussed including polychlorinated biphenyls, phthalate esters, nitrilotriacetate (NTA), numerous metals, and chlorine. The six Sections do not provide summaries; an understanding of how the recommendations should be interpreted and used can be gained only by a thorough reading of the rationale and the evaluation of criteria preceding the recommendations. (See also W74-[illegible])

*Water quality standards, *Reviews, *Regulation, *Baseline studies, Basic data collections, Recreation, Aesthetics, Water supply, Aquatic life, Wildlife, Irrigation water, Industrial water, Freshwater fish, Marine animals, Marine fish, *Toxicity, *Lethal limit, *Toxins, Bibliographies

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Water Quality Criteria 1972

A Report of the
Committee on Water Quality Criteria

Environmental Studies Board

National Academy of Sciences
National Academy of Engineering

Washington, D.C., 1972

At the request of
and funded by
The Environmental Protection Agency
Washington, D.C., 1972

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NOTICE

The study reported herein was undertaken under the aegis of the National Research Council with the express approval of the Governing Board of the NRC. Such approval indicated that the Board considered that the problem is of national significance, that elucidation or solution of the problem required scientific or technical competence, and that the resources of NRC were particularly suitable to the conduct of the project. The institutional responsibilities of the NRC were then discharged in the following manner:

The members of the study committee were selected for their individual scholarly competence and judgment with due consideration for the balance and breadth of disciplines. Responsibility for all aspects of this report rests with the study committee, to whom we express our sincere appreciation.

Although the reports of our study committees are not submitted for approval to the Academy membership nor to the Council, each report is reviewed by a second group of appropriately qualified individuals according to procedures established and monitored by the Academy's Report Review Committee. Such reviews are intended to determine, among other things, whether the major questions and relevant points of view have been addressed and whether the reported findings, conclusions, and recommendations arose from the available data and information. Distribution of the report is approved, by the President, only after satisfactory completion of this review process.

July 22, 1972

THE HONORABLE WILLIAM D. RUCKELSHAUS
Administrator
Environmental Protection Agency
Washington, D.C.

DEAR MR. RUCKELSHAUS:

It is our pleasure to transmit to you the report *Water Quality Criteria, 1972* prepared by the National Academy of Sciences-National Academy of Engineering Committee on Water Quality Criteria.

This book is the successor to the Water Quality Criteria Report of the National Technical Advisory Committee to the Secretary of the Interior in 1968. The 1972 Report drew significantly on its 1968 predecessor; nevertheless the current study represents a complete reexamination of the problems, and a critical review of all the data included here. The conclusions offered reflect the best judgment of the Academies' Committee.

The Report develops scientific criteria arranged in categories of major beneficial use. We are certain that the information and conclusions contained in this Report will be of use and value to the large number of people throughout the country who are concerned with achieving a high level of water quality for the Nation.

It is our pleasure to note the substantial personal contributions of the members of the Committee on Water Quality Criteria and its Panels and advisers. They have contributed more than 2,000 man-days of effort for which they deserve our gratitude. In less than a year and a half, they have collected a vast amount of scientific and technical information and presented it in a way that we believe will be most helpful to Federal and State officials as well as to the scientific community and the public. Oversight responsibility for the document, of course, rests with the Committee on Water Quality Criteria ably chaired by Dr. Gerard A. Rohlich of the University of Texas at Austin.

We wish also to express our appreciation to the Environmental Protection Agency which, without in any way attempting to influence the Committee's conclusions, provided technical expertise and information as well as the resources to undertake the study.

In the course of their work the Committee and Panels identified several scientific and technical areas in which necessary data is insufficient or lacking. The Academies find that a separate report is urgently required that specifies research needs to enable an increasingly effective evaluation of water quality. We are currently preparing such a report.

Sincerely yours,

PHILIP HANDLER
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PREFACE

In 1971, at the request of the ~~United States~~ Environmental Protection Agency, the National Academy of Sciences–National Academy of Engineering undertook the revision of WATER QUALITY CRITERIA, the 1968 Report of the National Technical Advisory Committee (NTAC) to the Secretary of the Interior. The Academies appointed a Committee on Water Quality Criteria and six Panels, and the responsibility for overseeing their activities was assigned to the Environmental Studies Board, a joint body of the Academies.

The guidelines for the Academies' Committee were similar to those followed by the NTAC. The Federal Water Pollution Control Act of 1948, as amended by the Water Quality Act of 1965, authorized the states and the federal government to establish water quality standards for interstate and coastal waters. Paragraph 3, Section 10 of the 1965 Act reads as follows:

> Standards of quality established pursuant to this subsection shall be such as to protect the public health or welfare, enhance the quality of water and serve the purposes of this Act. In establishing such standards the Secretary, the Hearing Board, or the appropriate state authority shall take into consideration their use and value for public water supplies, propagation of fish and wildlife, recreational purposes, and agricultural, industrial, and other legitimate uses.

Because of the vast amount of material that falls into the rubric of fish and wildlife, the Academies established separate Panels for freshwater and marine aquatic life and wildlife. Thus the Committee's six Panels were: (1) Recreation and Aesthetics, (2) Public Water Supplies, (3) Freshwater Aquatic Life and Wildlife, (4) Marine Aquatic Life and Wildlife, (5) Agricultural Uses of Water, and (6) Industrial Water Supplies.

The members of the Committee and its Panels were scientists and engineers expert and experienced in the various disciplines associated with the subject of water quality. The Panels also drew upon special advisors for specific water quality concerns, and in addition were aided by Environmental Protection Agency experts as liaison at the Panel meetings. This arrangement with EPA facilitated the Panels' access to EPA data on water quality. Thirty-nine meetings were held by the Committee and its Panels resulting in an interim report to the Academies and the Environmental Studies Board on December 1, 1971. This was widely circulated, and comments on it were solicited from many quarters. The commentaries were then considered for inclusion by the Committee and the appropriate Panels. This volume, submitted for publication in August 1972, within eighteen months of the inception of the task, is the final version of the Committee's report.

The 1972 Report ~~is vastly more than a revision of the NTAC Report. To begin with, it is nearly four times longer~~. Many new subjects are discussed in detail, among them: the recreational impact of boating, levels of use, disease vectors, nuisance organisms, and aquatic vascular plants; viruses in relation to public water supplies; effects of total dissolved gases on aquatic life; guidelines for toxicological research on pesticides and uses of toxicants in fisheries management; disposal of solid wastes in the ocean; use of waste water for irrigation; and industrial water treatment processes

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and resultant wastes. Many toxic or potentially toxic substances not considered by the NTAC are discussed including polychlorinated biphenyls, phthalate esters, nitrilotriacetate (NTA), numerous metals, and chlorine. The additional length also reflects the greater current awareness of how various characteristics of water affect its quality and use; and the expansion of the information base of the NTAC Report through new data from recent research activities and the greater capabilities of information processing, storage, and retrieval—especially evident in the three appendixes—have made their impact on the increase in size. In spite of these additions, however, the 1972 Report differs from the NTAC Report in that its six Sections do not provide summaries. The Committee agreed that an understanding of how the recommendations should be interpreted and used can be gained only by a thorough reading of the rationale and the evaluation of criteria preceding the recommendations.

Although each Section was prepared by its appropriate Panel, some discussions reflect the joint effort of two or more Panels. These combined discussions attempt to focus attention where desirable on such subjects as radioactivity, temperature, nutrient enrichment, and growths of nuisance organisms. However, the majority of topics were most effectively treated by individual Panel discussions, and the reader is encouraged to make use of the Tables of Contents and the index in assessing the full range of the Report's coverage of the many complex aspects of water quality.

Water quality science and its application have expanded rapidly, but much work remains to be done. In the course of this revision, the Committee and its Panels have identified many areas where further knowledge is needed, and these findings, now in preparation, will be published separately by the National Academy of Sciences–National Academy of Engineering as a report on research needs.

Social perspectives and policies for managing, enhancing, and preserving water resources are undergoing rapid and pervasive change. Because of the stipulations of the 1965 Water Quality Act, interstate water resources are currently categorized by use designation, and standards to protect those uses are developed from criteria. It is in this context that the Report of the NAS–NAE Committee, like that of the NTAC, was prepared. Concepts of managing water resources are subject to social, economic, and political decisions and will continue to evolve; but the Committee believes that the criteria and recommendations in this Report will be of value in the context of future as well as current approaches that might be taken to preserve and enhance the quality of the nation's water resources.

GERARD A. ROHLICH
Chairman, Committee on Water Quality Citeria

ACKNOWLEDGEMENTS

The NAS–NAE Committee on Water Quality Criteria and its Panels are grateful for the assistance of many institutions, groups, and individuals. The Environmental Studies Board provided guidance throughout all phases of the project, and the Environmental Protection Agency cooperated in making available their technical and informational resources. Many research organizations and individuals contributed unpublished data for the Panels' examination and use in the Committee's Report.

Numerous groups and individuals provided reviews and comments, among them several Federal agencies with staff expertise in water quality sciences, including NOAA of the Department of Commerce, the Department of the Interior, the Department of Health, Education, and Welfare, the Atomic Energy Commission, the Department of Agriculture, and the Department of the Army; scientists from the University of Wisconsin, especially Drs. Grant Cottam, G. Fred Lee, Richard B. Corey, David Armstrong, Gordon Chesters, Mr. James Kerrigan; and many others from academic, government, and private research institutions, including the National Research Council.

Much useful information including data and literature references was provided by the Water Resources Scientific Information Center of the U.S. Department of the Interior, the National Referral Center of the Library of Congress, the Defense Documentation Center of the U.S. Department of Defense, and the Library of the National Academy of Sciences–National Academy of Engineering. We are indebted to James L. Olsen, Jr. and Marilyn J. Urion of the Academies' Library and to Robert R. Hume, National Academy of Sciences Publications Editor, for their assistance.

Thanks are also due the many staff personnel of the Academies who assisted the project, particularly Linda D. Jones, Patricia A. Sheret, Eva H. Galambos, and Elizabeth A. Wilmoth. Mrs. Susan B. Taha was a tireless editorial assistant. The comprehensive author and subject indexes were prepared by Mrs. Bev Anne Ross.

Finally, each Committee and Panel member is indebted to his home institution and staff for their generous support of his efforts devoted to the Report of the Committee on Water Quality Criteria.

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GENERAL INTRODUCTION

HISTORICAL BACKGROUND

The past decade has been a period of unprecedented activity directed to man's concern for the quality of the environment, but a look at history shows that this concern, although currently intensified, is not new. The lessons of history and the findings of archaeologists provide concrete evidence that at least three thousand years before the birth of Christ man was cognizant of the need to dispose of his wastes and other refuse if he was to keep his environment livable.[1] For thousands of years the guidelines to quality of the water resource apparently were based on the senses of smell, sight, and taste. Whether or not these organoleptic observations on the suitability of water for use would match today's criteria is questionable in light of Reynolds' reference to "the old woman in the Fens" who "spoke for many besides herself when she asked of the new and pure supply: *Call ye that water?* For she said, *it has neither taste nor smell!*"[2]; or in light of the more recent decision of a state supreme court in 1904, which took the position that it is "not necessary to weigh with tenderness and care the testimony of experts . . . an ordinary mortal knows whether water is fit to drink and use."[3]

Although the concern for water quality is not new, progress has been made in moving from sensory associations as a means of control to the application of knowledge and criteria gained from scientific advances in detection and measurement, and in a greater understanding of the characteristics of water. Essentially it has been the developments of the past century that have provided criteria for and knowledge of water quality characteristics upon which we base determinations of its suitability for particular uses.

Until recently, relatively few scientists and engineers had been engaged in this field. The past decade, however, has seen a tremendous increase in the number of workers devoted to the subject of water quality assessment. Concurrently, an increasing awareness of the public has become apparent. As Leopold states, "The outstanding discovery of the twentieth century is not television, or radio, but rather the complexity of the land organism"; and he points out that "by land is meant all of the things on, over, or in the earth."[4] The growing public awareness of environmental quality has helped to accelerate activity directed to the solution of problems relating to water quality.

Forty centuries before the germ theory of disease had the support of scientifically conducted experiments, some control measures to provide safe water supplies were in use. Boiling, filtration through charcoal, and the practice of siphoning off water clarified by sedimentation were among the early methods used to improve water quality.[5] The regard of the Romans for high quality water is well known, and their civil works in obtaining water by the construction of aqueducts and the carrying away of waste waters in the cloacae or sewers, and in particular the Cloaca Maxima, are matters of common knowledge. The decline of sanitation through the Middle Ages and into the early part of the past century brought on the ravages of pestilence and the scourges of cholera, typhoid fever and dysentery, which led to the resurgence of public concern over water quality. There were many experiments and suggestions regarding filtration for purification as early as the 17th century. They culminated in design of the first filters for municipal supplies by Gibbs in Scotland in 1804 and in England in 1829 by Simpson who is probably most renowned for his work in constructing filters for the Chelsea Water Company to supply water for London from the Thames River.

The relationship of water quality to disease was firmly established by the report on the Broad Street Well in London by Sir John Snow in 1849, and in Edwin Chadwick's report of 1842 "On an inquiry into the Sanitary Condition of the Labouring Population of Gt. Britain."[6] The greatest part of Chadwick's report developed four major axioms that are still of relevance today. The first axiom established the cause and effect relationship between "insanitation, defective drainage, inadequate water supply, and overcrowded housing" on the one hand, and "disease, high mortality rates, and low expectation of life" on the other. The second axiom discussed the economic cost of ill health. The third dealt with the "social cost of squalor," and the fourth was concerned with the "inherent inefficiency of existing legal and administrative machinery." Chadwick argued that the "only hope of sanitary improvement lay in radical administrative departures" which would call for new institutional arrangements.

It is evident from these few glimpses into the early years of development of control that the basic approach, and justifiably so, was to provide water suitable for human use. A century ago the principal aim was to provide, by bacteriological examination, a scientific basis on which to establish water quality practices for protection of the public health. Increasingly, however, we have come to recognize that a multitude of materials that may occur in water have adverse effects on beneficial uses other than that for public water supplies.

WATER QUALITY CONTROL IN THE UNITED STATES

McKee and Wolf have provided an excellent historical background to the development of water quality standards and criteria and have summarized the water quality criteria promulgated by federal, state, and interstate agencies up to 1963.[7] Since then, many federal and state acts have been passed and modifications made in state administrative codes designed to establish criteria and standards. Of particular significance in this respect was the impact of the Federal Water Pollution Control Act of 1948[8] as amended by the Water Quality Act of 1965.[9] The latter required that the states adopt:

- water quality criteria applicable to interstate waters; and
- a plan for the implementation and enforcement of the water quality criteria adopted.

The Act further noted that the criteria and plans would, upon approval by the federal government, become the applicable water quality standards. At that time the Federal Water Pollution Control Administration was in the Department of Health, Education, and Welfare. In May of 1966, the FWPCA was transferred to the Department of the Interior, and in April, 1970 it was renamed The Federal Water Quality Administration. In December, 1970, interstate water quality and pollution control activities became the concern of the Environmental Protection Agency.

On April 1, 1968, the FWPCA published the report of the National Technical Advisory Committee to the Secretary of the Interior entitled *Water Quality Criteria.*[10] This report, often referred to as the "Green Book," contains recommendations on water quality criteria for various uses. The present volume is a revision of that work with the objective of compiling and interpreting the most recent scientific data in order to establish what is known about the materials present in water as related to specific uses.

MAJOR WATER USES AS AN ORGANIZING APPROACH

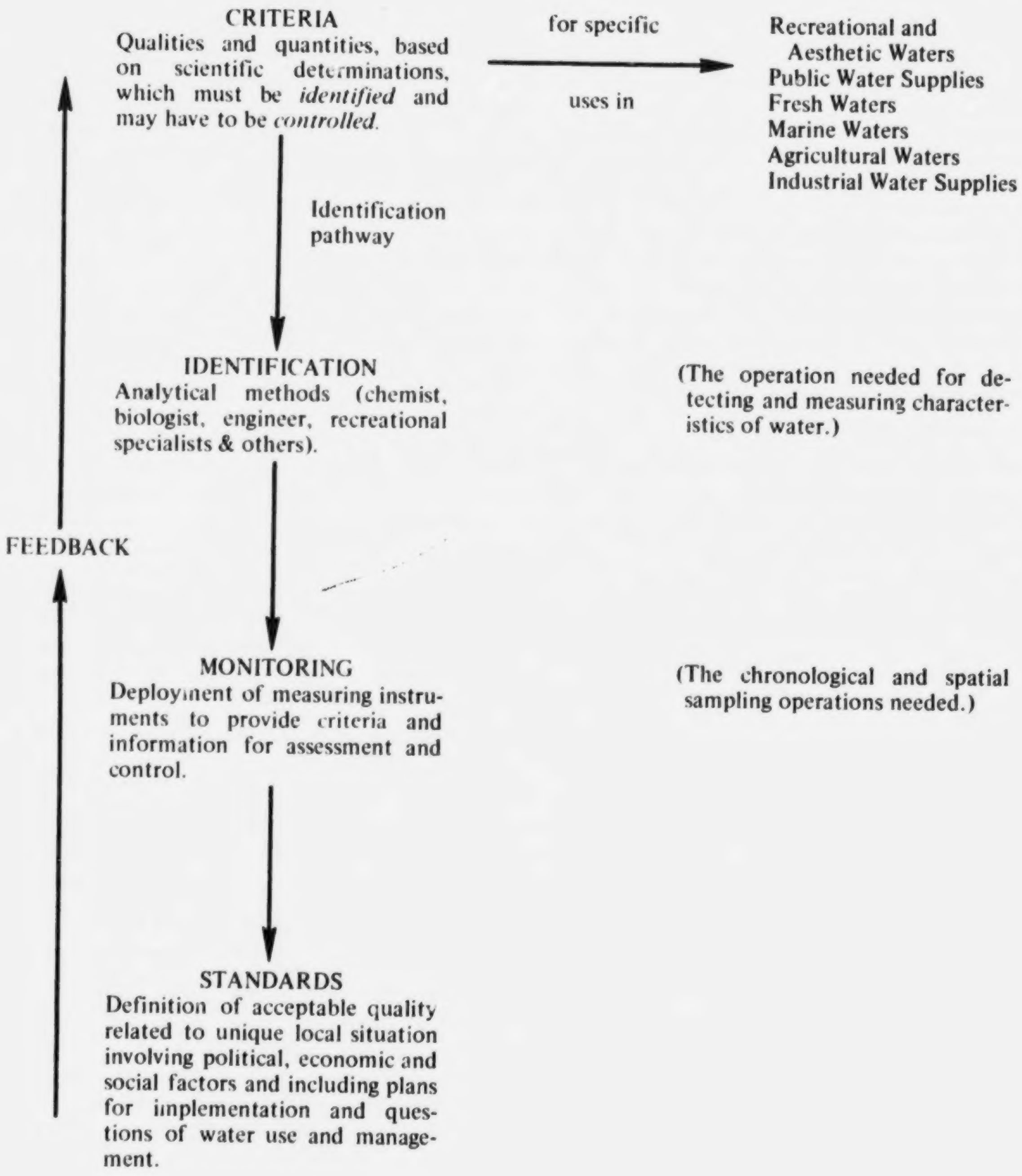
Although it is recognized that consideration must be given to the multiple use requirements placed on our water resources, this revision has followed the approach of the 1968 report in making recommendations in certain use categories. Such an approach provides a convenient way of handling an otherwise unwieldy body of data. Neither the approach itself nor the sequence in which the uses are arranged in the Report imply any comment on the relative importance of each use. Each water use plays its vital role in the water systems concept discussed above, and political, economic, and social considerations that vary with historical periods and geographic locations have brought particular water uses to positions of preeminent importance. In contemporary terms, it is not difficult to argue the primary importance of each water use considered in this Report: the recreational and aesthetic use of the Nation's water resources involves 3.7 billion man-days a year;[11] our public water supply systems prepare 15 billion gallons per day for the urban population alone;[12] commercial fishermen harvested 166,430,000 pounds of fish from the nation's public inland freshwater bodies in 1969;[13] our marine waters yield five billion pounds of fish annually for human use;[14] agriculture consumes 123 billion gallons of water per day in meeting its domestic, livestock, and irrigation needs;[15] and our industries must have 84,000 billion gallons of water per year to maintain their operations.[16]

Clearly, the designation of one water use as more vital than another is as impossible as it is unnecessary. Furthermore, we must not even restrict our thinking to present concepts and designated uses. Those concerned with water quality must envisage future uses and values that may be assigned to our water resources and recognize that man's activities in altering the landscape and utilizing water may one day have to be more vigorously controlled.

THE MEANING OF WATER QUALITY CRITERIA

In current practice, where multiple uses are required, as they will be in most situations, our guidelines to action will be the more stringent criteria. Criteria represent attempts to quantify water quality in terms of its physical, chemical, biological, and aesthetic characteristics. Those who are confronted with the problem of establishing or evaluating criteria must do so within the limits of the objective and subjective measurements available to them. Obviously, the quality of water as expressed by these measurements is the product of many changes. From the moment of its condensation in the atmosphere, water accumulates substances, in solution and suspension, from the air, from contacts as it moves over and into the land resource, from biological processes, and from human activities. Man affects the watershed as he alters the landscape by urbanization, by agricultural development, and by discharging municipal and industrial residues into the water resource. Thus climatic conditions, topography, geological formations, and human use and abuse of this vital resource significantly affect the characteristics of water, so that its quality varies widely with location and the influencing factors.

To look ahead again, it should be stressed that if coming generations expect to use future criteria established by



CRITERIA
Qualities and quantities, based on scientific determinations, which must be *identified* and may have to be *controlled.*

for specific uses in

Recreational and Aesthetic Waters
Public Water Supplies
Fresh Waters
Marine Waters
Agricultural Waters
Industrial Water Supplies

Identification pathway

IDENTIFICATION
Analytical methods (chemist, biologist, engineer, recreational specialists & others).

(The operation needed for detecting and measuring characteristics of water.)

FEEDBACK

MONITORING
Deployment of measuring instruments to provide criteria and information for assessment and control.

(The chronological and spatial sampling operations needed.)

STANDARDS
Definition of acceptable quality related to unique local situation involving political, economic and social factors and including plans for implementation and questions of water use and management.

FIGURE 1—Conceptual Framework for Developing Standards from Criteria

aquatic scientists, baseline areas must be preserved in which the scientists can work. Limnologists, oceanographers, and freshwater and marine biologists obtain baseline data from studies of undisturbed aquatic ecosystems. Because all the basic information has not yet been extracted from important study sites, it is essential that the natural condition of these sites prevail.

The fundamental point of departure in evaluating criteria for water quality in this Report is that the assignment of a level of quality is relative to the use man makes of that water. To evaluate the quality of water required for various uses, it is essential to know the limits of quality that have a detrimental effect on a designated use. As a corollary, in deciding whether or not water will be of suitable quality, one must determine whether or not the introduction into, or presence of any material in the resource, interferes with, alters, or destroys its intended use. Such decisions are subject to political, social, and economic considerations.

CRITERIA AND STANDARDS

The distinction between *criteria* and *standards* is important, and the words are not interchangeable nor are they synonyms for such commonly used terms as objectives or goals. As a clarification of the distinction that must be recognized and the procedural steps to be followed in developing standards from criteria, a conceptual framework based on the report "Waste Management and Control" by the Committee on Pollution NAS-NRC[17] is presented in Figure 1. In this context, the definition of criteria as used in this Report is "the scientific data evaluated to derive recommendations for characteristics of water for specific uses."

As a first step in the development of standards it is essential to establish scientifically based recommendations for each assignable water use. Establishment of recommendations implies access to practical methods for detecting and measuring the specified physical, chemical, biological, and aesthetic characteristics. In some cases, however, less than satisfactory methods are available, and in other cases, less than adequate methods or procedures are used. Monitoring the essential characteristics can be an operation concurrent with the identification step. If adequate criteria for recommendations are available, and the identification and monitoring procedures are sound, the fundamentals are available for the establishment of effective standards. It is again at this step that political, social, and economic factors enter into the decision-making process to establish standards.

Although the Committee and its Panels recognize that water quality, water quantity, water use, and waste water disposal form a complex system that is further complicated by the interchanges that occur among the land, air, and water resources, this Report cannot be so broad in scope: its explicit purpose is to recommend water quality characteristics for designated uses in light of the scientific information available at this time. We are aware that in some areas the scientific information is lacking, inadequate, or possibly conflicting thus precluding the recommendation of specific numerical values. The need to refine the recommendations and to establish new ones will become increasingly important as additional field information and research results become available. Realistic standards are dependent on criteria, designated uses, and implementation, as well as identification and monitoring procedures; changes in these factors may provide a basis for altering the standards.

Recommendations are usually presented, either as numerical values or in narrative form as summaries. In some instances in place of recommendations, conclusions based on the preceding discussion are given. It is important that each discussion be studied because it attempts to make clear the basis and logic used in arriving at the particular recommendation. The Committee wishes to emphasize the caveat so clearly stated in the introduction to the "Green Book." The Committee "does not want to be dogmatic" in making its recommendations. "They are meant as guidelines only, to be used in conjunction with a thorough knowledge of local conditions."[18]

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INTRODUCTION

This section considers water quality in the context of recreation and aesthetics, on the basis of available scientific data tempered by experience and judgment. In view of today's burgeoning population in the United States, the importance of water quality criteria to preserve and enhance the recreational and aesthetic values of water resources is manifest. The problems involved are both great and urgent. Our urban centers bear the brunt of the growth of a population that needs and demands water-oriented recreational resources. But those resources, already overloaded, are degraded or rendered unfit for recreation by the effects of man's activities. The quality of water can be assessed and to some extent controlled, but the principal cause of water pollution is what man does on the land. Water must be protected from harmful land-water relationships, and man must be protected from the consequences of degraded water quality.

THE ROLE OF WATER-ORIENTED RECREATION AND AESTHETICS

Recreation is an enigma: nearly everyone participates in some type of recreation, but few are likely to agree on an acceptable definition of it. Most persons who are not professionally involved with recreation tend to define it narrowly in terms of their own experiences. Many feel that the term implies some form of strenuous physical activity; to them, aesthetic appreciation and other leisure activities that primarily involve the mind are not "recreation." There is also a tendency for some to include only those physical activities that are commonly identified as "recreation" by public or quasi-public recreation agencies.

Charles E. Doell, an internationally known authority on park and recreation planning and administration, defines recreation as "the refreshment of the mind or body or both through some means which is in itself pleasureful." He states "almost any activity or mental process may be recreation depending largely upon the attitude assumed in the approach to the process itself" (Doell 1963)[4].* This concept is supported by many others (Brightbill 1961[2], Butler 1959[3], Lehman 1965[5]). If the attitude of the individual concerned is the key to whether or not an activity may be classed as "recreation," it follows that one man's work may be another man's recreation; and an unwelcome social duty to one person may be a valuable recreational experience to another. Certain activities may be either recreational or part of the daily routine depending on the attitude of the participant. Recreation is, therefore, an elusive concept that can bear some relationship to any of the major concerns of living—work and education, social duty, or bodily needs. Whether or not an individual's activity falls within the psychological realm of recreation depends upon his attitudes, goals, and life style at a point in time.

For the purposes of this report a broad view of recreation is adopted, and aesthetic appreciation is considered part of recreation. Thus the term "recreation" includes all types of intensive and extensive pleasurable activities ranging from sedentary, purely aesthetic experiences to strenuous activities that may involve a relatively small aesthetic component.

SCOPE AND NATIONAL SIGNIFICANCE

The scope and significance of water-related recreational activities is not well documented quantitatively, but an impression of its importance in the lives of Americans can be obtained from such evidence as license registration and sales data, user surveys, economic impact studies, and new legislation programs and regulations.

License Registration and Sales Data In 1960, 19 million persons bought 23 million state fishing licenses, tags, permits, and stamps. Ten years later more than 31 million licenses, tags, permits, and stamps were held by over 24.5 million purchasers, an increase of about 28 per cent over 1960 (U.S. Department of the Interior 1961,[13] 1971[14]). In 1970 sportsmen spent an estimated $287.7 million on fishing tackle and equipment on which they paid $14 million in federal excise taxes (Dingle-Johnson Act). They also added $90.9 million to state treasuries (Slater 1972),[7] and in many cases these funds were matched with federal funds for use in fisheries improvement programs.

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* Citations are listed at the end of the Section. They can be located alphabetically within subtopics or by their superior numbers which run consecutively across subtopics for the entire Section.

The number of recreational boats in use increased even more substantially. It was estimated that there were almost 9 million boats of various types in use during 1970, an increase of 9 per cent over 1966. More than $3 billion were spent at the retail level on boating equipment, services, insurance, fuel, mooring fees and memberships, a 22 per cent increase over 1966 (*The Boating Industry* 1971)[1]. In 1970, an estimated million pairs of water skis were sold, a 5 per cent increase in domestic and export sales for that year (*The Boating Industry* 1971)[1].

Economic Impact Studies In fiscal 1969–70, the Corps of Engineers spent $27.6 million to develop or expand facilities for swimming, fishing, boating, and other water-oriented activities (Stout *personal communication* 1971)[18]. The state parks of the nation, the majority of which are water-oriented, spent $125.8 million in 1970 on capital improvements and $177 million on operations and maintenance (Stout *personal communication* 1971)[18].

Although public expenditures for water-oriented recreational developments are large, expenditures in the private and commercial sectors are of even greater magnitude. In regions of the country where water bodies are reasonably numerous, most seasonal homes are built on or adjacent to water. In 1970, it was estimated that 150,000 seasonal homes were built at a cost of $1.2 billion (Ragatz 1971)[6]. Some waterfront locations have been extensively developed for a variety of public, private, and commercial recreational purposes. The lakes and lake frontage properties of the Tennessee Valley Authority alone were estimated to contain water-based recreational equipment and facilities worth $77 million and land-based facilities and improvements valued at $178 million in 1968 (Churchill *personal communication* 1972)[16].

Expenditures for other goods and services associated with water-oriented recreation are also a major factor in the economy. Boaters, fishermen, campers, picnickers, and others spend considerable sums on transportation, accommodations, and supplies. For example, preliminary data show that some 2.9 million waterfowl hunters spent an estimated $245 million during 25 million recreation days in 1970 (Slater *personal communication* 1971)[17]. The Tennessee Valley Authority estimated in 1967 that sports fishermen using its reservoirs spent some $42 million in order to harvest 7,000 to 10,000 tons of fish (Stroud and Martin 1968)[8].

User Surveys Since World War II, per capita participation in most types of recreational activities has increased even more rapidly than the preceding data indicate. Attendance at National Park Service areas rose from 133 million visits in 1966 to 172 million in 1970, an increase of 29 per cent. In the same period, visits to Corps of Engineers reservoirs increased 42 per cent to a total of 276 million. Comparable figures for the national forests were 151 million in 1966, rising 14 per cent to 173 million in 1970 (Bureau of Outdoor Recreation *personal communication* 1971)[15]. Most of the recreation opportunities at Corps of Engineers areas and a good proportion of those available on Park Service lands and in national forests are water-based or water-related. Similar growth rates and a predominance of water-related recreational experiences characterize the use of recreational lands managed by the Bureau of Sport Fisheries and Wildlife, the Bureau of Land Management, the Bureau of Reclamation, and the Department of Defense.

The preeminent role of water resources in recreation was emphasized by the President's Outdoor Recreation Resources Review Commission in 1960. Extensive surveys showed that most people seeking outdoor recreation (90 per cent of all Americans) sought it in association with water, as indicated by the preliminary figures in Table I–1, a study made as part of the 1970 U.S. Census (Slater 1972)[17]. Although it is impossible to estimate what proportion of the use reported by the survey was actually associated with water for those activities that are not water-based but are often water-related, the data nevertheless emphasize the magnitude of current participation in water-oriented recreation.

If no more than half the time spent on the frequently water-related activities was in fact associated with water, the total man days for water-based and water-related activities in 1970 would be at least 3.7 billion man days.

Participation in water-based and water-oriented recreation is likely to increase in the forseeable future. The Bureau of Outdoor Recreation (1967)[13] predicts that by the year 2000 summertime participation in swimming will increase over the year 1965 by 207 per cent, in fishing 78 per cent, in boating 215 per cent, in waterskiing 363 per cent, and in such water-related activities as camping, picnicking, and sightseeing 238, 127, and 156 per cent respectively.

Legislation, Regulations, and Programs The importance of water-based and water-related recreation to society is reflected in the increase in legislation and the number of regulations and programs intended to increase

TABLE I–1—Participation in Water-Oriented Recreation Activities in 1970

Activity	Percent of U.S. population participating[a]	Billions of man days
Water-based		
Swimming	46	1.72
Fishing	29	.56
Boating	24	.42
Total man days		2.70
Frequently water related		
Picnicking	49	.54
Birdwatching	4	.43
Camping	21	.40
Nature walks	18	.37
Hunting	12	.22
Wildlife photography	3	.04
Total man days		2.00

[a] For many activities, double counting will occur. (Slater 1972)[7]

or protect opportunities for these activities. One example is the Wild and Scenic Rivers Act U.S. Congress 1968)[9] that authorized a national program to preserve free-flowing river of exceptional natural or recreational value. The Federal Power Commission has required the submission of recreation and fish and wildlife development plans as integral parts of hydroelectric license applications. The Federal Water Project Recreation Act (U.S. Congress 1965)[10] encourages state and local participation in planning, financing, and administering recreational features of federal water development projects. The Estuary Protection Act (U.S. Congress 1968)[11] authorizes cooperative federal-state-local cost sharing and management programs for estuaries, and requires that federal agencies consult with the Secretary of the Interior on all land and water development projects with impacts on estuaries before submitting proposals to Congress for authorization.

The Soil Conservation Service of the U.S. Department of Agriculture assists in the development of ponds that often are used for recreational purposes and watering livestock. Federal assistance for waterfront restoration and the preservation of environmental values is available under the urban renewal, open space, and urban beautification programs of the Department of Housing and Urban Development, the Land and Water Conservation Fund program of the Bureau of Outdoor Recreation, and the historic preservation program of the National Park Service.

MAINTAINING AND RESTORING WATER QUALITY FOR RECREATION AND AESTHETICS

Although there have been instances of rapid water quality deterioration with drastic effects on recreation, typically the effect is a slow, insidious process. Changes have come about incrementally as forests are cut, land cultivated, urban areas expanded, and industries developed. But the cumulative effect and the losses in recreation opportunities caused by degraded water quality in this country in the past 100 years have been great. In many urban areas, opportunities for virtually every type of water-based activity have been either severely curtailed or eliminated. The resource-based recreation frontier is being forced further into the hinterland. Aesthetic values of aquatic vistas are eliminated or depreciated by enchroachment of residential, commercial, industrial, military, or transportation facilities. Drainage of swamps to control insect vectors of disease and channelization to control floods have a profound effect on water run-off characteristics. A loss in water quality and downstream aquatic environments and recreational opportunities is often the price paid for such improvements.

The application of adequate local, state, and national water quality criteria is only a partial solution to our water quality problems. A comprehensive national land use policy program with effective methods of decision-making, implementation, and enforcement is also needed.

APPLYING RECOMMENDATIONS

Throughout this report the recommendations given are to be applied in the context of local conditions. This caveat cannot be over emphasized, because variabilities are encountered in different parts of the country. Specific local recommendations can be developed now in many instances and more will be developed as experience grows. Numerical criteria pertaining to other beneficial water uses together with the recommendations for recreational and aesthetic uses provide guidance for water quality management.

WATER QUALITY FOR PRESERVING AESTHETIC VALUES

Aesthetics is classically defined as the branch of philosophy that provides a theory of the beautiful. In this Section attention will be focused on the aesthetics of water in natural and man-made environments and the extent to which the beauty of that water can be preserved or enhanced by the establishment of water quality recommendations.

Although perceptions of many forms of beauty are profoundly subjective and experienced differently by each individual, there is an apparent sameness in the human response to the beauties of water. Aesthetically pleasing waters add to the quality of human experience. Water may be pleasant to look upon, to walk or rest beside, or simply to contemplate. It may enhance the visual scene wherever it appears, in cities or in the wilderness. It may enhance values of adjoining properties, public or private. It may provide a focal point of pride in the community. The perception of beauty and ugliness cannot be strictly defined. Either natural or man-made visual effects may add or detract, depending on many variables such as distance from the observer or the composition and texture of the surroundings. As one writer has said when comparing recreational values with aesthetics, "Of probably greater value is the relaxation and mental well-being achieved by viewing and absorbing the scenic grandeur of the great and restless Missouri. Many people crowd the 'high-line' drives along the bluffs to view this mighty river and achieve a certain restfulness from the proximity of nature" (Porges et al. 1952)[19].

Similarly, aesthetic experience can be enhanced or destroyed by space relationships. Power boats on a two-acre lake are likely to be more hazardous than fun, and the water will be so choppy and turbid that people will hardly enjoy swimming near the shore. On the other hand, a sailboat on Lake Michigan can be viewed with pleasure. If a designated scenic area is surrounded by a wire fence, the naturalness is obviously tainted. If animals can only be viewed in restricted pens, the enjoyment is likely to be less than if they could be seen moving at will in their natural habitat.

MANAGEMENT FOR AESTHETICS

The management of water for aesthetic purposes must be planned and executed in the context of the uses of the land, the shoreline, and the water surfaces. People must be the ultimate consideration. Aesthetic values relate to accessibility, perspective, space, human expectations, and the opportunity to derive a pleasurable reaction from the senses.

Congress has affirmed and reaffirmed its determination to enhance water quality in a series of actions strengthening the federal role in water pollution control and federal support for water pollution control programs of state and local governments and industry. In a number of states, political leaders and voters have supported programs to protect or even restore water quality with aesthetics as one of the values.

The recognition, identification, and protection of the aesthetic qualities of water should be an objective of all water quality management programs. The retention of suitable, aesthetic quality is more likely to be achieved through strict control of discharges at the source than by excessive dependence on assimilation by receiving waters. Paradoxically, the values that aesthetically pleasing water provide are most urgently needed where pollution problems are most serious as in the urban areas and particularly in the central portions of cities where population and industry are likely to be heavily concentrated.

Unfortunately, one of the greatest unknowns is the value of aesthetics to people. No workable formula incorporating a valid benefit-to-cost ratio has yet been devised to reflect tangible and intangible benefits accruing to conflicting uses or misuses and the cost of providing or avoiding them. This dilemma could be circumvented by boldly stating that aesthetic values are worth the cost of achieving them. The present public reaction to water quality might well support this position, but efforts in this area have not yet proceeded far enough to produce values worthy of wide acceptance. (See Appendix I.)

BASIS OF RECOMMENDATIONS FOR AESTHETIC PURPOSES

All surface waters should be aesthetically pleasing. But natural conditions vary widely, and because of this a series of descriptive rather than numerical recommendations is made. The descriptions are intended to provide, in general terms, for the protection of surface waters from substances or conditions arising from other than natural sources that might degrade or tend to degrade the aesthetic quality of the water. Substances or conditions arising from natural sources may affect water quality independently of human activities. Human activities that augment degradation from natural sources, such as accelerated erosion from surface disturbances, are not considered natural. The recommendations are also intended to cover degradation from "discharges or waste," a phrase embracing undesirable inputs from all sources attributable to human activities whether surface flows, point discharges, or subsurface drainages.

The recommendations that follow are essentially finite criteria. The absence of visible debris, oil, scum, and other matter resulting from human activity is a strict requirement for aesthetic acceptability. Similarly, recommended values for objectionable color, odor, taste, and turbidity, although less precise, must be measured as no significant increase over background. Characteristics such as excessive nutrients and temperature elevations that encourage objectionable abundance of organisms, e.g., a bloom of blue-green algae resulting from discharge of a waste with a high nutrient content and an elevated temperature, must be considered.

These recommendations become finite when applied as intended in the context of natural background conditions. Specific numbers would add little to the usefulness of the descriptive recommendations because of the varying acuteness of sensory perception and because of the variability of substances and conditions so largely dependent on local conditions.

The phrase "virtually free" of an objectionable constituent as used in the recommendations implies the concept of freedom from the undesirable effects of the constituent but not necessarily freedom from the constituent itself. This recognizes the practical impossibility of complete absence and the inevitability of the presence of potential pollutants to some degree.

Recommendations

Surface waters will be aesthetically pleasing if they are virtually free of substances attributable to discharges or waste as follows:

- **materials that will settle to form objectionable deposits;**
- **floating debris, oil, scum, and other matter;**
- **substances producing objectionable color, odor, taste, or turbidity;**
- **substances and conditions or combinations thereof in concentrations which produce undesirable aquatic life.**

FACTORS INFLUENCING THE RECREATIONAL AND AESTHETIC VALUE OF WATER

The many factors that influence the recreational and aesthetic value of water may be broadly grouped in two imprecise and overlapping but useful categories: physical and biological. Physical factors include geography, management and land use practices, and carrying capacity. Biological factors involve the effects of nuisance organisms and eutrophication, the role of aquatic plants, species diversity, and the introduction of exotic species. In making water quality recommendations that will maintain recreational and aesthetic values of surface waters, it is necessary to understand the interrelationships between these factors and water quality. The discussions in this Section emphasize those interrelationships, but additional useful detail can be found in other Sections of this Report, i.e., Public Water Supplies (II), Marine Aquatic Life and Wildlife (IV), and Agriculture (V). Cross references direct the reader to other sources at appropriate points in this Section.

Physical Factors Recommendations applicable to water-related environmental goals may well define those constraints that must be imposed on man's land-based activities and upon his physical contact with water if the quality of water is to be maintained at a level suited to recreational use. This is especially true of aesthetic enjoyment of water, because pleasurable aesthetic experiences are related to water in its environmental setting and to its changing appearance caused by wind, light, and other natural phenomena.

Man-made impoundments have provided numerous opportunities for recreation that have not existed before, but their operation in some instances presents a paradox for recreational users. Often such reservoirs are located on the upper reaches of rivers where the natural setting is itself conducive to aesthetic recreational enjoyment; but because they are often multipurpose projects, their operation for water supply, seasonal provision of flood storage, daily provision of hydroelectric power, or even seasonal fluctuation for mosquito control will change the water surface elevation, leave barren banks exposed, or cause noticeable or transient disruptions of the otherwise natural appearing setting. Where the impoundment specifically provides a public water supply, concerned water works personnel, fearing degradation of the quality of the water stored for this purpose, may impose limitations on the scope of recreational opportunities. Thus, the full potential for recreational and aesthetic uses of water may well be curtailed somewhat by the operational schedule of a water body needed for other purposes, even if the quality of the stored water meets the stipulated water quality criteria.

Control of turbidity represents another environment-related problem, one that must often be dealt with in terms of somewhat subjective local considerations. Recommendations for turbidity limits are best expressed as percentage increases over natural background conditions. The wastewater treatment processes normally employed are intended to control suspended particles and associated problems. Steps can also be taken to minimize erosion of soil disturbed by agriculture, construction, logging, and other human activities. Turbidity from urban and rural areas can be reduced by ponding or other sedimentation facilities. Wherever possible, spoils from dredging of navigable waters should be disposed of on land or at water sites in such a way that environmental damage is minimized. If necessary dredging for new construction or channel maintenance is performed with caution, it will not have adverse effects on water quality. (Effects of physical manipulation of the environment are discussed further in Section III on Freshwater Aquatic Life and Wildlife.)

Biological Factors Two principal types of biological factors influence the recreational and aesthetic value of surface waters: those that endanger the health or physical comfort of people and animals, and those that render water aesthetically objectionable or unusable as a result of its overfertilization. The former include vector and nuisance organisms; the latter, aquatic growths of microscopic and macroscopic plants.

The discussion turns next to the physical factors of recreational carrying capacity and sediment and suspended materials, and then to the biological factors.

RECREATIONAL CARRYING CAPACITY

In both artificial impoundments and natural bodies of water the physical, chemical, and biological characteristics of the water itself are not the only factors influencing water-oriented recreation. Depreciation of the recreational value of water caused by high levels of use is a growing problem that can be solved only by management techniques that either create more extensive facilities or limit the types and amounts of use to predetermined desirable levels or carrying capacities.

The recreational resource carrying capacity concept is not new. Recreation land managers have used carrying capacity standards for decades, but such standards have generally been developed intuitively rather than experimentally. Dana (1957)[24] called for empirical research in this field to provide better guidelines for management of recreation resources. The National Recreation and Parks Association reported in 1969 that almost no research of this type had been completed and that standards for water-oriented recreational activities then in use exhibited a disturbingly wide range of values (Chubb 1969)[21]. Among investigations of the carrying capacity of water for recreational boating currently being made are those at North Carolina State University and Michigan State University (Ashton and Chubb 1971).[20] A comparative study of the canoeing and trout fishing capacity of four rivers is taking place in Michigan (Colburn, *personal communication* 1971)[27]. Lucas (1964)[25] reported on an on-going recreational carrying capacity study of the Boundary Waters Canoe Area.

Until a number of these investigations are completed, the true nature and complexity of the factors involved in recreational carrying capacity will not be known. However, in the case of many water-oriented activities it is apparent that social, psychological, and economic factors are involved, as well as the physical characteristics of the water body (Chubb and Ashton 1969)[22]. For example, boaters on heavily used lakes in Southeast Michigan represent a broad spectrum of behavioral patterns and attitudes. Fishermen generally dislike high-density use and are particularly annoyed by speeding boats that create waves. They believe such activities disturb the fish. Waterfront home and cottage owners abhor the noise and litter generated by owners of transient boats on trailers. On the other hand, many water skiers enjoy relatively crowded conditions because of the social aspects of the experience; and some cruiser and pontoon boat owners enjoy viewing the skiers from their boats. Thus the boating carrying capacity of these waters involves the relative proportions of the various kinds of uses taking place and the life styles, recreational goals, and social aspirations of the boaters. Carrying capacity becomes a function of the levels of satisfaction achieved by the participants (Ashton and Chubb 1971).[20]

Screw propellers of powerboats operating in shallow waters create currents that often suspend sediments. Powerboats can also produce wake waves that cause shore erosion and result in water turbulence. Marl-bottomed lakes and silty, relatively narrow rivers are especially susceptible to prolonged turbidity generated by such disturbances. In many cases, bank erosion has been so severe that speed limitations and wake-wave restrictions have had to be imposed.

The size and configuration of a water body influence its recreational use and carrying capacity. Large lakes with a low ratio of shoreline-to-surface area tend to be under-used in the middle; conversely, lakes with a high ratio of shoreline-to-surface area tend to sustain more recreational use per acre.

The Role of Regulation

Rapid increases in recreational use have necessitated regulations to protect the quality of the experiences obtained by limiting use so that carrying capacity is not exceeded. Examples are boat speed regulations, limitations on horsepower, number of boat launching sites, number of parking places, and zoning and time limitations on water skiing and high-speed boating. Motorized crafts are often prohibited. Michigan is planning to use data from its current series of boating carrying capacity studies to establish new criteria for its boating access site program (Ashton and Chubb 1971).[20]

The Michigan Department of Natural Resources (1970)[26] has proposed rationing recreation on stretches of the Au Sable, Manistee, Pine, and Pere Marquette Rivers by means of a canoe permit system to reduce conflicts between canoeists and trout fishermen. The proposed regulations would limit the release of canoes to a specified number per day for designated stretches of these rivers. Other regulations are intended to promote safety and reduce trespass, river bank damage, vandalism, and littering. The National Park Service has limited annual user days for river running on the Colorado through the Grand Canyon (Cowgill 1971).[23]

Factors Affecting Recreational Carrying Capacity

The carrying capacity of a body of water for recreation is not a readily identifiable finite number. It is a range of values from which society can select the most acceptable limits as the controlling variables change.

The schematic diagram (Fig. I-1) provides an impression of the number of relationships involved in a typical water body recreation system. Recreational carrying capacity of water is basically dependent upon water quality but also related to many other variables as shown in the model. At the threshold level a relatively small decline in water quality may have a considerable effect on the system and result in a substantial decline in the annual yield of water-oriented recreational opportunities at the sites affected.

Conclusion

No specific recommendation is made concerning recreational carrying capacity. Agencies establishing carrying capacities should be aware of the complex relationships of the interacting variables and of the constant need to review local established values in light of prevailing conditions. Carrying

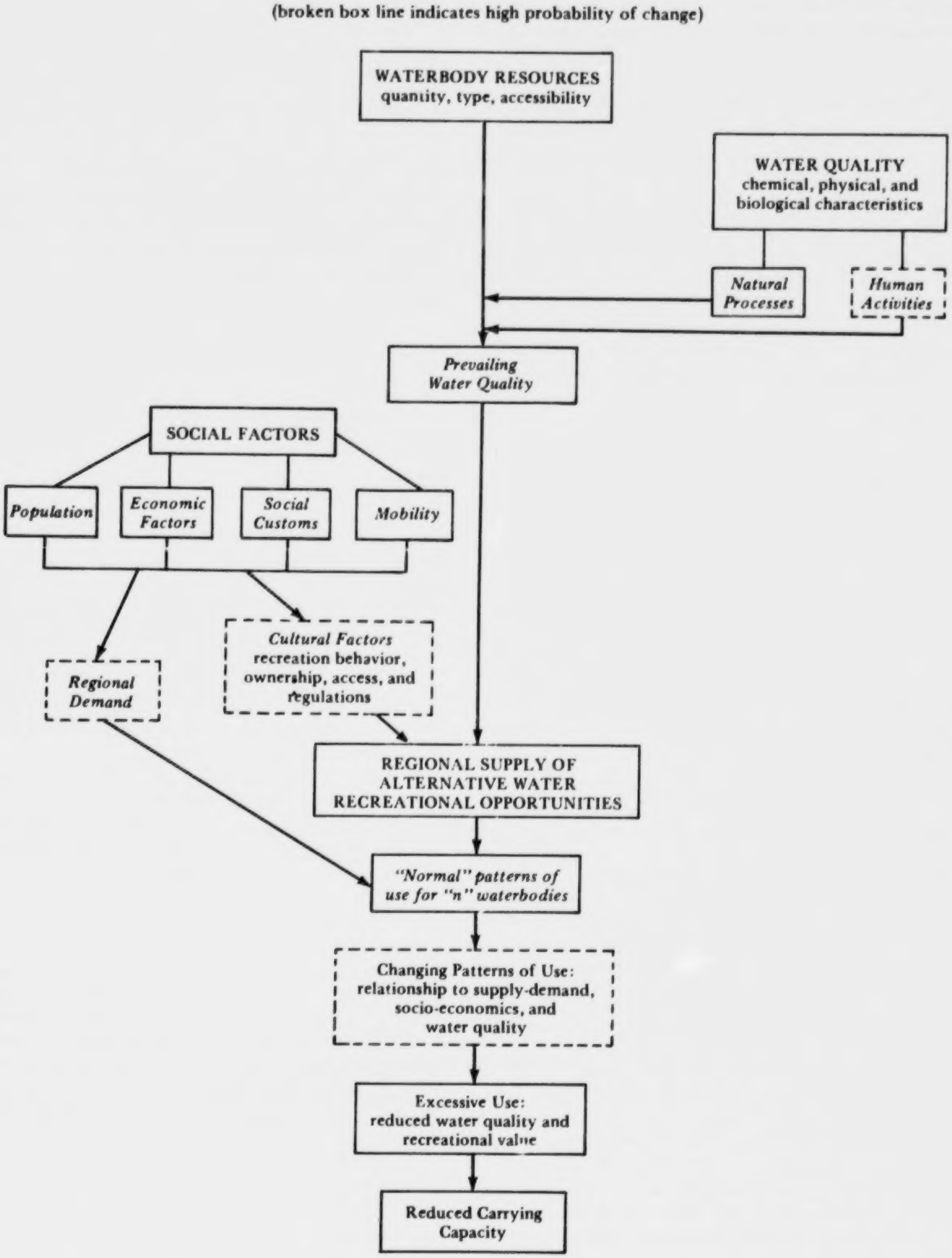
(broken box line indicates high probability of change)
WATERBODY RESOURCES
quantity, type, accessibility
WATER QUALITY
chemical, physical, and biological characteristics
Natural Processes
Human Activities
Prevailing Water Quality
SOCIAL FACTORS
Population
Economic Factors
Social Customs
Mobility
Regional Demand
Cultural Factors
recreation behavior, ownership, access, and regulations
REGIONAL SUPPLY OF ALTERNATIVE WATER RECREATIONAL OPPORTUNITIES
"Normal" patterns of use for "n" waterbodies
Changing Patterns of Use: relationship to supply-demand, socio-economics, and water quality
Excessive Use: reduced water quality and recreational value
Reduced Carrying Capacity

FIGURE I-1—Relationships Involved in a Water Resource Recreation System

capacity was discussed in this Section to call attention to its potential effects on water quality for recreational use.

SEDIMENTS AND SUSPENDED MATERIALS

Weathering of the land surface and the transport of particles such as sand, silt and clay by water, wind, and ice are natural processes of geologic erosion that largely determine the characteristics of our land, rivers, estuaries, and lakes. Man, however, can drastically alter the amount of material suspended in surface waters by accelerating surface erosion through various land use and management practices. Sources of these sediments and suspended materials such as erosion, mining, agriculture, and construction areas are discussed in Section IV on Marine Aquatic Life and Wildlife. In addition to causing siltation problems and affecting biological productivity, sediments and suspended materials affect the quality of surface waters used for recreational and aesthetic enjoyment.

Effects on Water Quality

The importance of suspended particle composition and concentrations to the recreational and aesthetic value of surface water relates to its effects on the clarity, light penetration, temperature, and dissolved constituents of surface water, the adsorption of toxic materials, and the composition, distribution, and rate of sedimentation of materials. These in turn not only affect recreational and aesthetic values directly, but they control or limit biological productivity and the aquatic life the waters will sustain for enjoyment by people (Buck 1956,[28] Cairns 1968).[29] Although the qualitative effects of suspended particles on surface waters are well recognized, quantitative knowledge and understanding are limited. (Biological effects are discussed in Sections III and IV on Freshwater and Marine Aquatic Life.)

Appearance The appearance of water is relative to the perspective of the viewer and his expectations. For example, the surfaces of lakes, streams, or oceans viewed from shore appear less turbid than they do viewed from above or during immersion. The responses of people viewing the spectacularly clear waters of Lake Tahoe or Crater Lake are almost surely aesthetic in nature, and allowing the clarity of such waters to decrease would certainly lower their aesthetic appeal. On the other hand, the roaring reaches and the placid stretches of the muddy Colorado River and miles of the muddy Mississippi afford another kind of aesthetic pleasure and recreation which many also appreciate. People seem to adapt to and accept a wide range of water turbidities as long as changes in turbidity are part of natural processes. However, increases in turbidity of water due to man's disturbance of the land surface, discharge of wastes, or modification of the water-body bed are subjectively regarded by many people as pollution, and so in fact or in fancy they reduce aesthetic enjoyment.

Light Penetration The presence of suspended solid materials in natural waters limits the penetration by sunlight. An example of the adverse effects of reduced available light is the inability of some fish to see their natural food or even the sport fisherman's lure (note the discussion in Section III, Freshwater Aquatic Life and Wildlife, pp. 126–129). In turbid, nutrient-rich waters, such as an estuary or lake where lack of light penetration limits algal reproduction, a water management project that reduced sediment input to the water body could conceivably result in increases of algal production to the nuisance level.

Temperature When suspended particles inhibit the penetration of water by sunlight, greater absorption of solar energy occurs near the surface and warms the water there. With its density thus decreased, the water column stabilizes, and vertical mixing is inhibited. Lower oxygen transfer from air to water also results from higher water surface temperature. Together with inhibited vertical mixing, this reduces the downward rate of oxygen transfer, especially in still or slowly moving water. In combination with the oxygen demand of benthic accumulations, any reduction in downward transfer of oxygen hastens the development of anaerobic conditions at the bed of shallow eutrophic ponds, and the result may be a loss of aesthetic quality.

Adsorption of Materials Clay minerals have irregular, platy shapes and large surface areas with electrostatic charges. As a consequence, clay minerals sorb cations, anions, and organic compounds. Pesticides and heavy metals likewise sorb on suspended clay particles, and those that are strongly held are carried with the particles to their eventual resting place.

Microorganisms are frequently sorbed on particulate material and incorporated into bottom sediments when the material settles. Rising storm waters may resuspend the deposited material, thereby restoring the microorganisms to the water column. Swimming or wading could stir bottom sediments containing bacteria, thereby effecting a rise in bacterial counts in the water (Van Donsel and Geldreich 1971)[33].

The capacity of minerals to hold dissolved toxic materials is different for each material and type of clay. The sorptive phenomenon effectively lends a large assimilative capacity to muddy waters. A reduction in suspended mineral solids in surface waters can, therefore, cause an increase in the concentrations of dissolved toxic materials contributed by existing waste discharges (see Section III on Freshwater Aquatic Life).

Beach Zone Effects When typical river waters containing dispersed clay minerals mix with ocean water in estuaries to the extent of one part or more of ocean water to 33 parts river water, the dispersed clay and silt particles become cohesive, and aggregates are formed under the prevailing hydraulic conditions (Krone 1962).[30] Such aggregates of material brought downstream by storms either settle in the estuary, particularly in large shallow bays, or are carried directly to sea where they often are distributed over large areas of the sea floor. Those that settle in shallow bays can be constantly resuspended by wind-generated waves and held in suspension by waves while tidal currents circulate the waters throughout the estuary and carry a portion of the suspended material out to sea. Suspended clay mineral particles are weakly cohesive in river waters having either unusually low dissolved salt concentrations or high proportions of multivalent cations in the dissolved salts. When such rivers enter lakes and impoundments, the fine particles aggregate and settle to the bed to form soft, fluffy deposits.

On lakes, the natural wind waves maintain beaches and sandy littoral zones when there is sufficient fetch. Wind-driven movement of the water through wave action and subsequent oscillation provides the minimum velocity of 0.5 feet per second to sort out the fine particles of mineral soils and organic micelles and allow them to settle in the depths. Wave action extends to depths of approximately one-half of the wave length to sort bottom sediments. This depth is on the order of 5 feet (1.5 m) for a one-mile (1.6 km) fetch. When the waters are deep enough to allow settling, fine sediments which are suspended drop down over the wave terrace leaving sorted sand behind. In shallow water bodies where the orbital velocity of the water particles of wave action is great enough to lift fine sediments, waters may be kept in a state of turbidity (Shephard 1963).[31] Waters without adequate wind-wave action and circulation do not have appreciable sorting; and therefore soft bottom materials, undesirable at facilities like swimming beaches, may build up in the shallows. These conditions reduce clarity and not only affect the aesthetic value but also present a hazard in swimming.

The natural phenomenon of beach maintenance, supplying sand to beaches and littoral zones, is dependent in part upon having ample sources of sand such as those provided by river transport and shore erosion. Impoundment of rivers causes sand to settle behind dams and removes it as a future source for beach maintenance. Man's protection of shorelines from erosion also interrupts the supply of sand. In the erosion process, sand is commonly moved along the shore in response to the net positive direction of the wind-wave forces, or it is carried into deep water to be deposited on the edge of wave terraces. The location of man-made structures can, therefore, influence the quality of beaches. Piers and jetties can intercept the lateral movement of sand and leave impoverished rocky or hardpan shores on the up-current side. Such conditions are common along the shores of the large Great Lakes and many coastal waters (U.S. Army, Coastal Engineering Research Center 1966).[32]

Sediment-Aquatic Plant Relationships When the sediment load exceeds the transport capacity of the river, deposition results. The accumulation of sediments in reservoirs and distribution systems has been a problem since ancient times. The deposited materials may so alter the original bed materials of surface waters that rooted aquatic vascular plants are able to grow in the newly available substrate, thus changing the aquatic environment. Fine sediments are often rich in the nutrients required for plant growths; and once the sediments are stabilized with a few plants, extensive colonization may follow. (See the discussion of Aquatic Vascular Plants in this Section.)

Recommendation

Clear waters are normally preferred for recreation. Because sediment-laden water reduces water clarity, inhibits the growth of plants, displaces water volume as sediments settle, and contributes to the fouling of the bottom, prevention of unnatural quantities of suspended sediments or deposit of sediments is desirable. Individual waters vary in the natural amounts of suspended sediments they carry; therefore, no fixed recommendation can be made. Management decisions should be developed with reference to historical base line data concerning the individual body of water.

VECTORS AND NUISANCE ORGANISMS

The impact of both aquatic vectors of diseases and nuisance organisms on water-related recreational and aesthetic pursuits varies from the creation of minor nuisances to the closing of large recreational areas (Mackenthun and Ingram 1967).[58] Organisms of concern are discussed by Mackenthun (1969).[57]

Massive emergences of non-biting midges, phantom midges, caddisflies, and mayflies cause serious nuisances in shoreline communities, impeding road traffic, river navigation, commercial enterprises and recreational pursuits (Burks 1953,[40] Fremling 1960a,[46] 1960b;[47] Hunt and Bischoff 1960;[54] Provost 1958[60]). Human respiratory allergic reactions to aquatic insect bites have been recognized for many years. They were reviewed by Henson (1966),[49] who reported the major causative groups to be the caddisflies, mayflies, and midges.

Among common diseases transmitted by aquatic invertebrates are encephalitis, malaria, and schistosomiasis, including swimmers' itch. The principal water-related arthropod-borne viral disease of importance to public health in the United States is encephalitis, transmitted by mosquitoes (Hess and Holden 1958).[51] Many polluted urban streams are ideally suited to production of large numbers of *Culex fatigans*, a vector of St. Louis encephalitis in urban areas. Although running waters ordinarily are not suitable for mosquito breeding, puddles in drying stream beds and floodplains are excellent breeding sites for this and other species of *Culex*. If such pools contain polluted waters, organic materials present may serve as an increased food supply that will stimulate production (Hess 1956,[50] U.S.

Department of the Interior, FWPCA 1967).[65] Aquatic plants also provide breeding sites for some mosquitoes and other nuisance insects. This relationship is discussed elsewhere in this Section (p. 25).

Other than mosquitoes, perhaps the most common nuisance insects associated with standing freshwater are chironomid midges. These insects neither bite nor carry disease, but their dense swarms can interfere with man's comfort and activities. Nuisance populations have occurred in productive natural lakes where the larvae thrive in the largely organic bottom sediments (Provost 1958,[60] Hunt and Bischoff 1960,[54] Hilsenhoff 1959).[52] In poorly designed sewage lagoons mosquitoes and midges may thrive (Beadle and Harmstrom 1958,[38] Kimerle and Enns 1968).[56] Reservoirs receiving inadequately treated municipal wastes are potential sources for abundant mosquito and midge production (U.S. Department of the Interior, FWPCA 1967).[65] Increased midge production may be associated with deterioration in water quality, but this is not always the case. For example, excessive production can occur in primary sewage oxidation ponds as well as in reservoirs (Grodhaus 1963,[48] Bay 1964[35]); and in sequential oxidation pond treatment, maximum midge production may sometimes occur in those ponds furthest from the plant effluent where water quality is highest (Bay et al. 1965).[36]

Abrupt changes in water quality such as dilution of seawater by freshwater, especially if accompanied by organic loading, can precipitate extraordinarily high midge production (Jamnback 1954).[55] Sudden decline in oxygen supply in organically overloaded ponds or drying lakes can disrupt or destroy established faunal communities, thus favoring midge larvae because they are tolerant to low dissolved oxygen and are primarily detrital feeders (Bay *unpublished data*).[67]

The physical characteristics of certain water bodies, as much as their water quality characteristics, may sometimes determine midge productivity (Bay et al. 1966).[37] For example, freshly filled reservoirs are quickly sedimented with allocthanous detritus and airborne organic matter that provide food for invading midge larvae. The rate of sedimentation can depend on watershed characteristics and basin percolation rate or, in the case of airborne sediment, on the surrounding topography. Predators in these new environments are few, and initial midge larval survival is high. Thomas (1970)[64] has also reported on the potential of newly or periodically flooded areas to produce large populations of midges and mosquitoes.

Midge production in permanent bodies of water is extremely variable. Attempts have been made (Hilsenhoff and Narf 1968,[53] Florida State Board of Health *unpublished data*[69]) to correlate factors of water quality with midge productivity in neighboring lakes and in lakes with certain identifiable characteristics, but the results have been inconclusive.

Organism response in organically polluted flowing water was discussed and illustrated by Bartsch and Ingram (1959).[34] As water quality and bottom materials change in streams recovering from organic waste discharges, large numbers of midges and other nuisance organisms may be produced in select reaches.

Though blackfly larvae are common in unpolluted streams, an increase in suspended organic food particles may stimulate increased populations, and abnormally large numbers of larvae have been found downstream from both municipal and industrial waste discharges (U.S. Department of the Interior, FWPCA 1967).[65] The larvae feed on drifting organic material, and either municipal, agricultural, or certain industrial wastes can provide the base for an increased food supply. Bacteria from soils and sewage may be important in outbreaks of blackflies (Fredeen 1964).[46]

Toxic wastes can also affect situations where nuisance organisms are found in increased numbers. The most obvious mechanism is the destruction of more sensitive predators and competitors, leaving the food supply and space available for the more tolerant forms. Surber (1959)[63] found increased numbers of a tolerant midge, *Cricotopus bicinctus*, in waters polluted with chromium. Rotenone treatment of waters has resulted in temporary massive increases in blackfly and midge populations (Cook and Moore 1969).[41] Increased numbers of midge larvae were found in a stream reach six months after a gasoline spill (Bugbee and Walter 1972).[68] The reasons for this are not clear but may be linked to the more ready invasion of an area by these highly mobile insects as compared to less mobile competitors and predators.

Persons involved in water-based activities in many areas of the world are subject to bilharziasis (schistosomiasis), a debilitating and sometimes deadly disease (World Health Organization 1959).[66] This is not a problem in the continental United States and Hawaii because of the absence of a vector snail, but schistosomiasis occurs in Puerto Rico due to the discharge of human feces containing *Schistosoma* eggs into waters harboring vector snails, the most important species being *Biomphalaria glabrata*. *B. glabrata* can survive in a wide range of water quality, including facultative sewage lagoons; and people are exposed through contact with shallow water near the infected snails. Cercariae shed by the snail penetrate the skin of humans and enter the bloodstream.

Of local concern in water-contact recreation in the United States is schistosome dermatitis, or swimmers' itch (Cort 1928,[42] Mackenthun and Ingram 1967,[58] Fetterolf et al. 1970).[44] A number of schistosome cercariae, nonspecific for humans, are able to enter the outer layers of human skin. The reaction causes itching, and the severity is related to the person's sensitivity and prior exposure history (Oliver 1949).[59] The most important of the dermatitis-producing cercariae are duck parasites (*Trichobilharzia*).

Snails serving as intermediate hosts include *Lymnaea, Physa,* and *Gyraulus* (Cort 1950).[43] Although swimmers' itch has wide distribution, in the United States it is principally endemic to the north central lake region. Occasional incidence is reported in marine waters (Stunkard and Hinchliffe 1952).[62]

About 90 per cent of severe swimmers' itch outbreaks are associated with *Cercaria stagnicolae* shed from varieties of the snail *Lymnaea emarginata.* This relationship is promoted by (1) clean, sandy beaches ideal for swimming and preferred by the snail; (2) peak populations of the snail host that develop in sandy-bottomed lakes of glacial origin; (3) the greatest development of adult snails that do not die off until toward the end of the bathing season; and (4) the cycle of cercarial infection so timed that the greatest numbers of cercariae emerge during the hot weather in the middle of the summer when the greatest amount of bathing is done (Brackett 1941).[39] Infected vector snails are also found throughout the United States in swamps, muddy ponds, and ditches; but dermatitis rarely results, because humans seldom use these areas without protective clothing.

In some marine recreational waters jellyfish or sea nettles are serious problems. Some species possess stinging mechanisms whose cnidoblast filaments can penetrate human skin causing painful, inflammed weals. The effects of water quality on their abundance is not known, but Schultz and Cargo (1971)[61] reported that the summer sea nettle, *Chrysaora quinquecirrha,* has been a problem in Chesapeake Bay since colonial days. When these nettles are abundant, swimming is practically eliminated and fishermen's nets and traps are clogged.

Conclusion

The role of water quality in either limiting or augmenting the production of vector and nuisance organisms involves many interrelationships which are not clearly understood. Since organic wastes generally directly or indirectly increase biomass production, there may be an attendant increase in vector or nuisance organisms. Some wastes favor their production by creating water quality or habitat conditions that limit their predators and competitors. Increased production of vector and nuisance organisms may degrade a healthy and desirable human environment and be accompanied by a lessening of recreational and aesthetic values (see the discussion of Aquatic Life and Wildlife in this Section, p. 35.)

EUTROPHICATION AND NUTRIENTS

Man's recent concern with eutrophy relates primarily to lakes, reservoirs, rivers, estuaries, and coastal waters that have been or are being over-fertilized through society's carelessness to a point where beneficial uses are impaired or threatened. With increasing urbanization, industrialization, artificial soil fertilization, and soil mantle disruption, eutrophication has become a serious problem affecting the aesthetic and recreational enjoyment of many of the nation's waters.

Defining Eutrophication and Nutrients

Lakes have been classified in accordance with their trophic level or bathymetry as eutrophic, oligotrophic, mesotrophic, or dystrophic (National Academy of Sciences 1969,[97] Russell-Hunter 1970,[105] Warren 1971,[114] Stewart and Rohlich 1967).[107] A typical eutrophic lake has a high surface-to-volume ratio, and an abundance of nutrients producing heavy growth of aquatic plants and other vegetation; it contains highly organic sediments, and may have seasonal or continuous low dissolved-oxygen concentrations in its deeper waters. A typical oligotrophic lake has a low surface-to-volume ratio, a nutrient content that supports only a low level of aquatic productivity, a high dissolved-oxygen concentration extending to the deep waters, and sediments largely inorganic in composition. The characteristics of mesotrophic lakes lie between those of eutrophic and oligotrophic lakes. A dystrophic lake has waters brownish from humic materials, a relatively low pH, a reduced rate of bacterial decomposition, bottom sediments usually composed of partially decomposed vegetation, and low aquatic biomass productivity. Dystrophication is a lake-aging process different from that of eutrophication. Whereas the senescent stage in eutrophication may be a productive marsh or swamp, dystrophication leads to a peat bog rich in humic materials but low in productivity.

Eutrophication refers to the addition of nutrients to bodies of water and to the effects of those nutrients. The theory that there is a natural, gradual, and steady increase in external nutrient supply throughout the existence of a lake is widely held, but there is no support for this idea of natural eutrophication (Beeton and Edmondson 1972).[74] The paleolimnological literature supports instead a concept of trophic equilibrium such as that introduced by Hutchinson (1969).[91] According to this concept the progressive changes that occur as a lake ages constitute an ecological succession effected in part by the change in the shape of the basin brought about by its filling. As the basin fills and the volume decreases, the resulting shallowness increases the cycling of available nutrients and this usually increases plant production.

There are many naturally eutrophic lakes of such recreational value that extensive efforts have been made to control their overproduction of nuisance aquatic plants and algae. In the past, man has often accepted as a natural phenomenon the loss or decreased value of a resource through eutrophication. He has drained shallow, senescent lakes for agricultural purposes or filled them to form building sites. The increasing value of lakes for recreation, however, will reorder man's priorities, and instead of accepting such alternative uses of lakes, he will divert his reclamation efforts to salvaging and renovating their recreational values.

Artificial or cultural eutrophication results from increased nutrient supplies through human activity. Many aquatic systems have suffered cultural eutrophication in the past 50 years as a consequence of continually increasing nutrient loading from the wastes of society. Man-induced nutrients come largely from the discharge of municipal and industrial wastewaters and from the land runoff effects of agricultural practices and disruption of the soil mantle and its vegetative cover in the course of land development and construction. If eutrophication is not to become the future major deterrent to the recreational and aesthetic enjoyment of water, it is essential that unnatural additions of nutrients be kept out of water bodies through improved wastewater treatment and land management.

Effects of Eutrophication and Nutrients

Green Lake, a lowland lake with high recreation use in Seattle, is an example of a natural eutrophic lake (Sylvester and Anderson 1960),[109] formed some 25,000 years ago after the retreat of the Vashon glacier. During the ensuing years, about two-thirds of the original lake volume was filled with inorganic and organic sediments. A core taken near the center of the lake to a sediment depth of 20.5 feet represented a sediment accumulation over a period of approximately 6,700 years. Organic, nutrient, and chlorophyll analyses on samples from the different sediment depths indicated a relatively constant rate of sedimentation, suggesting that Green Lake has been in a natural state of eutrophy for several thousands of years.

The recreational and aesthetic potential of the lake was reduced for most users by littoral and emergent vegetation and by heavy blooms of blue-green algae in late summer. The aquatic weeds provided harborage for production of mosquitoes and interfered with boating, swimming, fishing, access to the beach, and model boat activities. The heavy, blue-green algal blooms adhered to swimmers. The wind blew the algal masses onto the shore where they decomposed with a disagreeable odor. They dried like a blue-green paint on objects along the shoreline, rendered boating and fishing unattractive, and accentuated water line marks on boats.

Nevertheless, through the continuous addition of low-nutrient dilution water by the City of Seattle (Oglesby 1969),[98] Green lake has been reclaimed through a reversal of the trophic development to mesotrophic and is now recreationally and aesthetically acceptable.

Lake Washington is an example of a large, deep, oligotrophic-mesotrophic lake that turned eutrophic in about 35 years, primarily through the discharge of treated and untreated domestic sewage. Even to laymen, the change was rapid, dramatic, and spectacular. In the period of a year, the apparent color of the lake water turned from bluish-green to rust as a result of massive growths of the blue-green alga, *Oscillatoria rubescens*. This threat to aesthetic and recreational enjoyment was a key factor in voter approval of Metro, a metropolitan sewer district. Metro has greatly reduced the nutrient content of the lake and consequent algal growth by diverting wastewater discharges out of the drainage basin (Edmondson 1969,[82] 1970).[83]

Lake Sammamish at the northern inlet of Lake Washington appeared to be responding to the enrichment it received from treated sewage and other nutrient waste, although it had not yet produced nuisance conditions to the extent found in Lake Washington (Edmondson 1970).[83] However, subsequent diversion of that waste by Metro has resulted in little or no detectable recovery in three years, a period that proved adequate for substantial recovery in Lake Washington (Emery et al. 1972).[85] Lake Sebasticook, Maine, affords another example of undesirable enrichment Although previously in an acceptable condition, it became obnoxious during the 1960's in response to sewage and a wide variety of industrial wastes (HEW 1966).[112] The nutrient income of Lake Winnisquam, New Hampshire, has been studied to determine the cause of nuisance blooms of blue-green algae (Edmondson 1969).[82] The well-known lakes at Madison, Wisconsin, including Monona, Waubesa, and Mendota, have been the object of detailed studies of nutrient sources and their deteriorating effect on water quality (Sawyer 1947,[106] Mackenthun et al. 1960,[95] Edmondson 1961,[80] 1968).[81]

A desirable aspect of eutrophication is the ability of mesotrophic or slightly eutrophic lakes typically to produce greater crops of fish than their oligotrophic or nutrient-poor counterparts. As long as nuisance blooms of algae and extensive aquatic weed beds do not hinder the growth of desirable fish species or obstruct the mechanics and aesthetics of fishing or other beneficial uses, some enrichment may be desirable. Fertilization is a tool in commercial and sport fishery management used to produce greater crops of fish. Many prairie lakes in the east slope foothills of the Rocky Mountains would be classed as eutrophic according to the characteristics discussed below, yet many of these lakes are exceptional trout producers because of the high natural fertility of the prairie (Sunde et al. 1970).[108] As an example of an accepted eutrophic condition, their waters are dense with plankton, but few would consider reducing the enrichment of these lakes.

Streams and estuaries, as well as lakes, show symptoms of over-enrichment, but there is less opportunity for buildup of nutrients because of the continual transport of water. Although aquatic growths can develop to nuisance proportions in streams and estuaries as a result of over-enrichment, manipulation of the nutrient input can modify the situation more rapidly than in lakes.

Man's fertilization of some rivers, estuaries, and marine embayments has produced undesirable aquatic growths of algae, water weeds, and slime organisms such as *Cladophora*, *Ulva*, *Potamogeton*, and *Sphaerotilus*. In addition to interfering with other uses, as in clogging fishing nets with slime (Lincoln and Foster 1943),[94] the accompanying water-quality changes in some instances upset the natural fauna and flora and cause undesirable shifts in the species composition of the community.

Determination of Trophic Conditions

It should be emphasized that (a) eutrophication has a significant relationship to the use of water for recreational and aesthetic enjoyment as well as the other water uses discussed in this book; (b) this relationship may be desirable or undesirable, depending upon the type of recreational and aesthetic enjoyment sought; and (c) the possible disadvantages or advantages of eutrophication may be viewed subjectively as they relate to a particular water use. There are no generally accepted guidelines for judging whether a state of eutrophy exists or by what criteria it may be measured, such as production of biomass, rate of productivity, appearance, or change in water quality. Ranges in primary productivity and oxygen deficit have been suggested as indicative of eutrophy, mesotrophy, and oligotrophy by Edmondson (1970)[83] and Rodhe (1969),[104] but these ranges have had no official recognition.

The trophic state and natural rate of eutrophication that exists, or would exist, in the absence of man's activities is the basis of reference in judging man-induced eutrophication. The determination of the natural state in many water bodies will require the careful examination of past data, referral to published historical accounts, recall by "old-timers," and perhaps the examination of sediment cores for indicator species and chemical composition. The following guidelines are suggested in determining the reference trophic states of lakes or detecting changes in trophic states. Determination of the reference trophic state accompanied by studies of the nutrient budget may reveal that the lake is already in an advanced state of eutrophy. For temperate lakes, a significant change in indicator communities or a significant increase in any of the other four indices, detectable over a five-year period or less, is considered sufficient evidence that accelerated eutrophication is occurring. An undetectable change over a shorter period would not necessarily indicate a lack of accelerated eutrophication. A change detectable only after five years may still indicate unnaturally accelerated eutrophication, but five years is suggested as a realistic maximum for the average monitoring endeavor. Where cultural eutrophication is suspected and changes in indices are not observable, analysis of sediment cores may be necessary to establish the natural state. The dynamic characteristics and individuality of lakes may produce exceptions to these guidelines. They are not infallible indicators of interference with recreation, but for now they may serve as a beginning, subject to modification as more complete data on the range of trophic conditions and their associated effects become available.

Primary Productivity Ranges in the photosynthetic rate, measured by radioactive carbon assimilation, have been suggested by Rodhe (1969)[104] as indicative of trophic conditions (Table I–2).

Biomass Chlorophyll *a* is used as a versatile measure of algal biomass. The ranges presented for mean summer chlorophyll *a* concentration determined in epilimnetic water supplies collected at least biweekly and analyzed according to *Standard Methods* (American Public Health Assoc., American Water Works Assoc., and Water Pollution Control Federation 1971)[70] are indices of the trophic stage of a lake: oligotrophic, 0–4 mg chlorophyll a/m^3; eutrophic, 10–100 mg chlorophyll a/m^3.

These ranges are suggested after reviewing data on chlorophyll concentrations and other indicators of trophic state in several lakes throughout the United States and Canada. Of greatest significance are data from Lake Washington which show that during peak enrichment, mean summer chlorophyll *a* content rose to about 27 mg/m^3 and that the lake was definitely eutrophic. The post nutrient diversion summer mean declined to about 7 mg/m^3, and the lake is now more typically mesotrophic (Edmondson 1970;[83] chlorophyll *a* values corrected to conform to recent analytical techniques). Unenriched and relatively low productive lakes at higher elevations in the Lake Washington drainage basin show mean summer chlorophyll *a* contents of 1 to 2 mg/m^3. Moses Lake, which can be considered hypereutrophic, shows a summer mean of 90 mg/m^3 chlorophyll *a* (Bush and Welch 1972).[76]

Oxygen Deficit Criteria for rate of depletion of hypolimnetic oxygen in relation to trophic state were reported by Mortimer (1941)[96] as follows:

oligotrophic	*eutrophic*
<250 mg $O_2/m^2/day$	>550 mg $O_2/m^2/day$

This is the rate of depletion of hypolimnetic oxygen determined by the change in mean concentration of hypolimnetic oxygen per unit time multiplied by the mean depth of the hypolimnion. The observed time interval should be at least a month, preferably longer, during summer stratification.

TABLE I–2—Ranges in Photosynthetic Rate for Primary Productivity Determinations[a]

Period	Oligotrophic	Eutrophic
Mean daily rates in a growing season, $mgC/m^2/day$	30–100	300–3000
Total annual rates, $gC/m^2/year$	7–75	75–700

[a] Measured by total carbon uptake per square meter of water surface per unit of time. Productivity estimates should be determined from at least monthly measurements according to *Standard Methods*.

American Public Health Association, American Water Works Assoc., and Water Pollution Control Federation 1971[70]; Rodhe 1969.[104]

Indicator Communities The representation of certain species in a community grouping in fresh water environments is often a sensitive indicator of the trophic state. Nutrient enrichment in streams causes changes in the size of faunal and floral populations, kinds of species, and numbers of species (Richardson 1928,[103] Ellis 1937,[84] Patrick 1949,[99] Tarzwell and Gaufin 1953[110]). For example, in a stream typical of the temperate zone in the eastern United States degraded by organic pollution the following shifts in aquatic communities are often found: in the zone of rapid decomposition below a pollution source, bacterial counts are increased; sludgeworms (Tubificidae), rattail maggots (*Eristalis tenax*) and bloodworms (Chironomidae) dominate the benthic fauna; and blue-green algae and the sewage fungus (*Sphaerotilus*) become common (Patrick 1949,[99] Tarzwell and Gaufin 1953,[110] Patrick et al. 1967[100]). Various blue-green algae such as *Schizothrix calcicola, Microcoleus vaginatus, Microcystis aeruginosa,* and *Anabaena* sp. are commonly found in nutrient-rich waters, and blooms of these and other algae frequently detract from the aesthetic and recreational value of lakes. Diatoms such as *Nitzschia palea, Gomphonema parvulum, Navicula cryptocephala, Cyclotella meneghiniana,* and *Melosira varians* are also often abundant in nutrient-rich water (Patrick and Reimer 1966).[101] Midges, leeches, blackfly larvae, *Physa* snails, and fingernail clams are frequently abundant in the recovery zone.

Nutrients Chemicals necessary to the growth and reproduction of rooted or floating flowering plants, ferns, algae, fungi, or bacteria are considered to be nutrient chemicals. All these chemicals are not yet known, but those that have been identified are classified as macronutrients, trace elements or micronutrients, and organic nutrients. The macronutrients are calcium, potassium, magnesium, sodium, sulfur, carbon and carbonates, nitrogen, and phosphorus. The micronutrients are silica, manganese, zinc, copper, molybdenum, boron, titanium, chromium, cobalt, and perhaps vanadium (Chu 1942,[77] Arnon and Wessell 1953,[72] Hansen et al. 1954).[89] Examples of organic nutrients are biotin, B_{12}, thiamine, and glycylglycine (Droop 1962).[79] Some of the amino acids and simple sugars have also been shown to be nutrients for heterotrophs or partial heterotrophs.

Plants vary as to the amounts and kinds of nutrients they require, and as a result one species or group of species of algae or aquatic plants may gain dominance over another group because of the variation in concentration of nutrient chemicals. Even though all the nutrients necessary for plant growth are present, growth will not take place unless environmental factors such as light, temperature, and substrate are suitable. Man's use of the watershed also influences the sediment load and nutrient levels in surface waters (Leopold et al. 1964,[93] Bormann and Likens 1967).[75]

Thomas (1953)[111] found that the important factor in artificial eutrophication was the high phosphorus content of domestic wastes. Nitrogen became the limiting growth factor if the algal demand for phosphorus was met. Numerous studies have verified these conclusions (American Society of Limnology and Oceanography 1972).[71]

Sawyer (1947)[106] determined critical levels of inorganic nitrogen (300 μg/l N) and inorganic phosphorus (10 μg/l P) at the time of spring overturn in Wisconsin lakes. If exceeded, these levels would probably produce nuisance blooms of algae during the summer. Nutrient concentrations should be maximum when measured at the spring overturn and at the start of the growing season. Nutrient concentrations during active growth periods may only indicate the difference between amounts absorbed in biomass (suspended and settled) and the initial amount biologically available. The values, therefore, would not be indicative of potential algal production. Nutrient content should be determined at least monthly (including the time of spring overturn) from the surface, mid-depth, and bottom. These values can be related to water volume in each stratum, and nutrient concentrations based on total lake volume can be derived.

One of the most convincing relationships between maximum phosphate content at the time of lake overturn and eutrophication as indicated by algal biomass has been shown in Lake Washington (Edmondson 1970).[83] During the years when algal densities progressed to nuisance levels, mean winter PO_4–P increased from 10–20 μg/l to 57 μg/l. Following diversion of the sewage mean PO_4–P decreased once again to the preenrichment level. Correlated with the PO_4–P reduction was mean summer chlorophyll *a* content, which decreased from a mean of 27 μg/l at peak enrichment to less than 10 μg/l, six years after diversion was initiated.

Although difficult to assess, the rate of nutrient inflow more closely represents nutrient availability than does nutrient concentration because of the dynamic character of these nonconservative materials. Loading rates are usually determined annually on the basis of monthly monitoring of water flow, nutrient concentration in natural surface and groundwater, and wastewater inflows.

Vollenweider (1968)[113] related nutrient loading to mean depths for various well-known lakes and identified trophic states associated with induced eutrophication. These findings showed shallow lakes to be clearly more sensitive to nutrient income per unit area than deep lakes, because nutrient reuse to perpetuate nuisance growth of algae increased as depth decreased. From this standpoint nutrient loading was a more valid criterion than nutrient concentration in judging trophic state. Examples of nutrient loadings which produced nuisance conditions were about 0.3 g/m^2/yr P and 4 g/m^2/yr N for a lake with a mean depth of 20 meters, and about 0.8 g/m^2/yr P and 11 g/m^2/yr N for a lake with a mean depth of 100 meters.

These suggested criteria apply only if other requirements of algal growth are met, such as available light and water retention time. If these factors limit growth rate and the increase of biomass, large amounts of nutrients may move through the system unused, and nuisance conditions may not occur (Welch 1969).[116]

Carbon (C) is required by all photosynthetic plants. It may be in the form of CO_2 in solution, HCO_3^-, or $CO_3^=$. Carbamine carboxylate, which may form by the complexing of calcium or other carbonates and amino compounds in alkaline water, is an efficient source of CO_2 (Hutchinson 1967).[90] Usually carbon is not a limiting factor in water (Goldman et al. 1971).[88] However, King (1970)[92] estimated that concentrations of CO_2 less than 3 micromoles at equilibrium favored blue-green algae, and concentrations greater than this favored green algae.

Cations such as calcium, magnesium, sodium, and potassium are required by algae and higher aquatic plants for growth, but the optimum amounts and ratios vary. Furthermore, few situations exist in which these would be in such low supply as to be limiting to plants. Trace elements either singly or in combination are important for the growth of algae (Goldman 1964).[86] For example molybdenum has been demonstrated to be a limiting nutrient in Castle Lake. Deficiencies in trace elements are more likely to occur in oligotrophic than in eutrophic waters (Goldman 1972).[87]

The vitamins important in promoting optimum growth in algae are biotin, thiamin, and B_{12}. All major groups require one or more of these vitamins, but particular species may or may not require them. As Provasoli and D'Agostino (1969)[102] pointed out, little is known about the requirement for these vitamins for growth of algae in polluted water.

Under natural conditions it is difficult to determine the effect of change in concentrations of a single chemical on the growth of organisms. The principal reasons are that growth results from the interaction of many chemical, physical, and biological factors on the functioning of an organism; and that nutrients arise from a mixture of chemicals from farm, industrial, and sanitary wastes, and runoff from fields. However, the increase in amounts and types of nutrients can be traced by shifts in species forming aquatic communities. Such biotic shifts have occurred in western Lake Erie (Beeton 1969).[73] Since 1900 the watershed of western Lake Erie has changed with the rapidly increasing human population and industrial development, as a result of which the lake has received large quantities of sanitary, industrial, and agricultural organic wastes. The lake has become modified by increased concentrations of dissolved solids, lower transparency, and low dissolved oxygen concentration. Blooms of blue-green algae and shifts in invertebrate populations have markedly increased in the 1960's (Davis 1964,[78] Beeton 1969).[73]

Summary of Measurement of Nutrient Enrichment

Several conditions can be used to measure nutrient enrichment or its effects:

- a steady decrease over several years in the dissolved oxygen content of the hypolimnion when measured prior to fall overturn, and an increase in anaerobic areas in the lower portion of the hypolimnion;
- an increase in dissolved materials, especially nutrients such as nitrogen, phosphorus, and simple carbohydrates;
- an increase in suspended solids, especially organic materials;
- a shift in the structure of communities of aquatic organisms involving a shift in kinds of species and relative abundances of species and biomass;
- a steady though slow decrease in light penetration;
- an increase in organic materials and nutrients, especially phosphorus, in bottom deposits;
- increases in total phosphorus in the spring of the year.

Recommendations

The principal recommendations for aesthetic and recreational uses of lakes, ponds, rivers, estuaries, and near-shore coastal waters are that these uses continue to be pleasing and undiminished by effects of cultural activities that increase plant nutrients. The trophic level and natural rate of eutrophication that exists, or would exist, in these waters in the absence of man's activities is considered the reference level and the commonly desirable level to be maintained. Such water should not have a demonstrable accelerated production of algae growth in excess of rates normally expected for the same type of waterbody in nature without man-made influences.

The concentrations of phosphorus and nitrogen mentioned in the text as leading to accelerated eutrophication were developed from studies for certain aquatic systems: maintenance of lower concentrations may or may not prevent eutrophic conditions. All the factors causing nuisance plant growths and the level of each which should not be exceeded are not known. However, nuisance growths will be limited if the addition of all wastes such as sewage, food processing, cannery, and industrial wastes containing nutrients, vitamins, trace elements, and growth stimulants are carefully controlled and nothing is added that causes a slow overall decrease of average dissolved oxygen concentration in the hypolimnion and an increase in the extent and duration of anaerobic conditions.

AQUATIC VASCULAR PLANTS

Aquatic vascular plants affect water quality, other aquatic organisms, and the uses man makes of the water. Generally, the effects are inversely proportional to the volume of the water body and directly proportional to the use man wishes to make of that water. Thus the impact is often most significant in marshes, ponds, canals, irrigation ditches, rivers, shallow lakes, estuaries and embayments, public water supply sources, and man-made impoundments. Dense growths of aquatic vascular plants are not necessarily due to human alteration of the environment. Where an appropriate environment for plant growth occurs, it is extremely difficult to prevent the growth without changing the environment. Addition of plant nutrients can cause aquatic vascular plants to increase to nuisance proportions in waters where natural fertility levels are insufficient to maintain dense populations (Lind and Cottam 1969).[147] In other waters where artificial nutrient additions are not a problem, natural fertility alone may support nuisance growths (Frink 1967).[135]

Interrelationships With Water Quality

Through their metabolic processes, manner of growth, and eventual decay, aquatic vascular plants can have significant effects on such environmental factors as dissolved oxygen and carbon dioxide, carbonate and bicarbonate alkalinity, pH, nutrient supplies, light penetration, evaporation, water circulation, current velocity, and sediment composition. The difficulty in understanding the interrelationships among plant growth and water quality is described in part by Lathwell et al. (1969).[144] Diurnal oxygen rhythm with maximum concentrations in the afternoon and minimums just before dawn is a universally-recognized limnological phenomenon, and metabolic activities of vascular plants can contribute to these rhythms. The effect of aquatic plants on dissolved oxygen within a reach of stream at a particular time of day is a function of the plant density and distribution, plant species, light intensity, water depth, turbidity, temperature, and ambient dissolved oxygen. Oxygen production is proportional to plant density only to a certain limit; when this limit is exceeded, net oxygen production begins to decrease and, with increasing density, the plants become net oxygen consumers (Owens et al. 1969).[159] It is hypothesized that this phenomenon occurs because the plants become so dense that some are shaded by other overlying plants. Westlake (1966)[173] developed a model for predicting the effects of aquatic vascular plant density and distribution on oxygen balance which demonstrates that if the weeds are concentrated within a small area, the net effect of the weeds may be to consume more oxygen than that produced, even though the average density may be relatively low.

After reviewing the literature on the direct effects of plants on the oxygen balance, Sculthorpe (1967)[162] concluded that the extent of oxygen enrichment at all sites varies with changing light intensity, temperature, and plant population density and distribution. On a cloudy, cool day community respiration may exceed even the maximum photosynthetic rate. Although vigorous oxygen production occurs in the growing season, the plants eventually die and decay, and the resulting oxygen consumption is spread over the cooler seasons of the year.

Light penetration is significantly reduced by dense stands of aquatic vascular plants, and this reduces photosynthetic rates at shallow depths. Buscemi (1958)[129] found that under dense beds of *Elodea* the dissolved oxygen concentration fell sharply with depth and marked stratification was produced. Severe oxygen depletion under floating mats of water hyacinth (Lynch et al. 1947),[150] duckweed and water lettuce (Yount 1963)[170] have occurred. Extensive covers of floating or emergent plants shelter the surface from the wind, reduce turbulence and reaeration, hinder mixing, and promote thermal stratification. Dense growths of phytoplankton may also shade-out submerged macrophytes, and this phenomenon is used to advantage in fisheries pond culture. Fertilization of ponds to promote phytoplankton growth is recommended as a means of reducing the standing crop of submerged vascular plants (Swingle 1947,[167] Surber 1961[166]).

Interrelationships of plants with water chemistry were reported by Straskraba (1965)[165] when foliage of dense populations of *Nuphar*, *Ceratophyllum*, and *Myriophyllum* were aggregated on the surface. He found pronounced stratification of temperature and chemical factors and reported that the variations of oxygen, pH, and alkalinity were clearly dependent on the photosynthesis and respiration of the plants. Photosynthesis also involves carbon dioxide, and Sculthorpe (1967)[162] found that for every rise of 2 mg/l of dissolved oxygen the total carbon dioxide should drop 2.75 mg/l and be accompanied by a rise in the pH. A rise in pH will allow greater concentrations of un-ionized ammonia (see Freshwater Aquatic Life, p. 140).

Hannan and Anderson (1971)[137] studied diurnal oxygen balance, carbonate and bicarbonate alkalinity and pH on a seasonal basis in two Texas ponds less than 1 m deep which supported dense growths of submerged rooted macrophytes. One pond received seepage water containing free carbon dioxide and supported a greater plant biomass. This pond exhibited a diurnal dissolved-oxygen range in summer from 0.8 to 16.4 mg/l, and a winter range from 0.3 to 18.0 mg/l. The other pond's summer diurnal dissolved-oxygen range was 3.8 to 14.9 mg/l and the winter range was 8.3 to 12.3 mg/l. They concluded that (a) when macrophytes use bicarbonate as a carbon source, they liberate carbonate and hydroxyl ions, resulting in an increase in pH and a lowered bicarbonate alkalinity; and (b) the pH of a macrophyte community is a function of the carbon dioxide-bicarbonate-carbonate ionization phenomena as altered by photosynthesis and community respiration.

Dense colonies of aquatic macrophytes may occupy up to 10 per cent of the total volume of a river and reduce the maximum velocity of the current to less than 75 per cent of that in uncolonized reaches (Hillebrand 1950,[139] as reported by Sculthorpe 1967[162]). This can increase sediment deposition and lessen channel capacity by raising the substrate, thus increasing the chance of flooding. Newly deposited silt may be quickly stabilized by aquatic plants, further affecting flow.

Loss of water by transpiration varies between species and growth forms. Otis (1914)[158] showed that the rate of transpiration of *Nymphaea odorata* was slightly less than the rate of evaporation from a free water surface of equivalent area, but that of several emergent species was up to three times greater. Sculthorpe (1967)[162] postuiated that transpiration from the leaves of free-floating rosettes could be at rates six times greater than evaporation from an equivalent water surface. Loss of water through water hyacinth was reported by Das (1969)[133] at 7.8 times that of open water.

Interrelationships With Other Biota

Aquatic macrophytes provide a direct or indirect source of food for aquatic invertebrates and fish and for wildlife. The plants provide increased substrate for colonization by epiphytic algae, bacteria, and other microorganisms which provide food for the larger invertebrates which, in turn, provide food for fish. Sculthorpe (1967)[162] presented a well-documented summary of the importance of a wide variety of aquatic macrophytes to fish, birds, and mammals. Sago pondweed (*Potamogeton pectinatus*) illustrates the opposite extreme in man's attitude toward aquatic macrophytes: Timmons (1966)[168] called it the most noxious plant in irrigation and drainage ditches of the American west, whereas Martin and Uhler (1939)[155] considered it the most important duck food plant in the United States.

Aquatic vegetation and flotage breaking the water surface enhance mosquito production by protecting larvae from wave action and aquatic predators and interfering with mosquito control procedures. Two major vectors of malaria in the United States are *Anopheles quadrimaculatus* east of the Rocky Mountains, and *A. freeborni* to the west (Carpenter and La Casse 1955).[130] Anopheline mosquitoes are generally recognized as permanent pool breeders. The more important breeding sites of these two mosquitoes are freshwater lakes, swamps, marshes, impoundment margins, ponds, and seepage areas (Carpenter and La Casse 1955).[130] The role of various aquatic plant types in relation to the production and control of *A. quadrimaculatus* on artificial ponds and reservoirs indicates that the greatest problems are created by macrophytes that are (1) free-floating, (2) submersed and anchored but which break the water surface, (3) floating leaf anchored, and (4) emersed floating-mat anchored (U.S. Department of Health, Education, and Welfare, Public Health Service, and Tennessee Valley Authority 1947).[169] In addition to vector mosquitoes, pestiferous mosquitoes develop in association with plant parts in shoreline areas. Jenkins (1964)[142] provided an annotated list and bibliography of papers dealing with aquatic vegetation and mosquitoes.

Generally, submersed vascular plants have lower nutrient requirements than filamentous algae or phytoplankton (Mulligan and Baranowski 1969).[157] Plants with root systems in the substrate do not have to compete with phytoplankton, periphyton, or non-rooted macrophytes for the phosphorus in the sediments.

Boyd (1971b),[126] relating his earlier work on emergent species (Boyd 1969,[122] 1970a,[123] 1971a[125]) to that of Stake (1967,[163] 1968[164]) on submerged species, stated that in the southern United States most of the total net nutrient accumulation by aquatic vascular plants occurs by midspring before peak dry matter standing crop is reached, and that nutrients stored during early spring growth are utilized for growth later. Thus nutrients are removed from the environment early in the season, giving the vascular hydrophytes a competitive advantage over phytoplankton. Boyd (1967)[121] also reported that the quantity of phosphorus in aquatic plants frequently exceeds that of the total water volume. These phenomena may account for the high productivity in terms of macrophytes which can occur in infertile waters. However, if the dissolved phosphorus level is not a limiting factor for the phytoplankton, the ability to utilize sediment phosphorus is not a competitive advantage for rooted plants.

Further interaction between aquatic vascular plants and phytoplankton has been demonstrated recently in studies showing that concentrations of dissolved organic matter can control plant growth in lakes by regulating the availability of trace metals and other nutrients essential to plant photosynthesis. An array of organic-inorganic interactions shown to suppress plant growth in hardwater lakes (Wetzel 1969,[174] 1971[175]) appear to operate in other lake types and streams (Breger 1970,[127] Malcolm et al. 1970,[152] Allen 1971[116]). Wetzel and Allen in press (1971)[176] and Wetzel and Manny (1972)[177] showed that aquatic macrophytes near inlets of lakes can influence phytoplankton growth by removing nutrients as they enter the lake while at the same time producing dissolved organic compounds that complex with other nutrients necessary to phytoplankton growth. Manny (1971,[153] 1972[154]) showed several mechanisms by which dissolved organic nitrogen (DON) compounds regulate plant growth and rates of bacterial nutrient regeneration. These control mechanisms can be disrupted by nutrients from municipal and agricultural wastes and dissolved organic matter from inadequately treated wastes.

Effects on Recreation and Aesthetics

It is difficult to estimate the magnitude of the adverse effects of aquatic macrophytes in terms of loss of recreational opportunities or degree of interference with recreational pursuits. For example, extensive growths of aquatic macrophytes interfere with boating of all kinds; but the extent of interference depends, among other things, on the growth form of the plants, the density of the colonization, the fraction of the waterbody covered, and the purposes, attitudes, and tolerance of the boaters. Extremes of opinion on the degree of impact create difficulty in estimating a monetary, physical, or psychological loss.

Dense growths of aquatic macrophytes are generally objectionable to the swimmer, diver, water skier, and scuba enthusiast. Plants or plant parts can be at least a nuisance to swimmers and, in extreme cases, can be a factor in drowning. Plants obstruct a diver's view of the bottom and underwater hazards, and fronds can become entangled in a scuba diver's gear. Water skiers' preparations in shallow water are hampered by dense growths of plants, and fear of falling into such growths while skiing detracts from enjoyment of the sport.

Rafts of free-floating plants or attached plants which have been dislodged from the substrate often drift onto beaches or into swimming areas, and time and labor are entailed in restoring their attractiveness. Drying and decaying aquatic plants often produce objectionable odors and provide breeding areas for a variety of insects.

Sport fishermen have mixed feelings about aquatic macrophytes. Fishing is often good around patches of lily pads, over deeply-submerged plants, and on the edges of beds of submerged weeds which rise near the surface. On the other hand, dense growths may restrict the movement and feeding of larger fish and limit the fishable area of a waterbody. Aquatic plants entangle lures and baits and can prevent fishermen from reaching desirable fishing areas.

Marshes and aquatic macrophytes in sparse or moderate densities along watercourse and waterbody margins augment nature study and shoreline exploration and add to the naturalistic value of camping and recreation sites. It is only when the density of the growths, or their growth forms, become a nuisance and interfere with man's activities that he finds them objectionable. An indication of how often that occurs is provided by McCarthy (1961),[156] who reported that on the basis of a questionnaire sent to all states in 1960, there were over 2,000 aquatic vegetation control projects conducted annually, and that most states considered excessive growth of aquatic vegetation a serious and increasing problem.

The aesthetic value of aquatic macrophytes is in the mind of the beholder. The age-old appeal of aquatic plants is reflected in their importance as motifs in ancient architecture, art, and mythology. Aquatic gardens continue to be popular tourist attractions and landscaping features, and wild aquatic plant communities have strong appeal to the artist, the photographer, and the public. To many, these plants make a contribution of their own to the beauty of man's environment.

Control Considerations

Aquatic vascular plants can be controlled by several methods: chemical (Hall 1961,[136] Little 1968[148]); biological (Avault et al. 1968,[117] Maddox et al. 1971,[151] Blackburn et al. 1971[120]); mechanical (Livermore and Wunderlich 1969[149]); and naturalistic environmental manipulation (Penfound 1953).[160] General reviews of control techniques have been made by Holm et al. (1969),[141] Sculthorpe (1967),[162] and Lawrence (1968).[146]

Harvesting aquatic vascular plants to reduce nutrients as a means of eutrophication control has been investigated by Boyd (1970b),[124] Yount and Crossman (1970),[171] and Peterson (1971).[161] Although many investigators have reported important nutrients in various aquatic plants, the high moisture content of the vegetation as it is harvested has been an impediment to economic usefulness. Peterson (1971)[161] reported the cost per pound of phosphorus, nitrogen, and carbon removed from a large lake supporting dense growths of aquatic vascular plants as $61.19, $8.24 and $0.61 respectively.

Nevertheless, improved methods of harvesting and processing promise to reduce the costs of removing these bothersome plants and reclaiming their nutrients for animal and human rations or for soil enrichment. Investigation into the nutritive value of various aquatic plants has frequently been an adjunct of research on the efficiency and economy of harvesting and processing these plants in an effort to remove nuisance growth from lakes and streams. Extensive harvesting of aquatic vegetation from plant-clogged Caddo Lake (Texas-Louisiana) was followed by plant analysis and feeding trials. The dehydrated material was found to be rich in protein and xanthophyll (Creger et al. 1963,[132] Couch et al. 1963[131]). Bailey (1965)[118] reported an average of 380 milligrams of xanthophyll per pound of vacuum oven-dried aquatic plant material with about 19 per cent protein. Hentges (1970),[138] in cooperation with Bagnall (1970),[119] in preliminary tests with cattle fed press-dehydrated aquatic forage, found that pelleted *Hydrilla verticillata* (Florida elodea) could be fed satisfactorily as 75 per cent of a balanced ration. Bruhn et al. (1971)[128] and Koegel et al. (1972)[143] found 44 per cent mineral and 21 per cent protein composition in the dry matter of the heat coagulum of the expressed juice of Eurasian water milfoil (*Myriophyllum spicatum*). The press residue, further reduced by cutting and pressing to 16 per cent of the original volume and 32 per cent of the original weight, could readily be spread for lawn or garden mulch.

Control measures are undertaken when plant growth interferes with human activities beyond some ill-defined point, but too little effort has been expended to determine the causes of infestations and too little concern has been given the true nature of the biological problem (Boyd 1971b).[126] Each aquatic macrophyte problem under consideration for control should be treated as unique, the biology of the plant should be well understood, and all the local factors thoroughly investigated before a technique is selected. Once aquatic macrophytes are killed, space for other plants becomes available. Nutrients contained in the original plants are released for use by other species. Long-term control normally requires continued efforts. Herbicides may be directly toxic to fish, fish eggs, or invertebrates important as fish food (Eipper 1959,[134] Walker 1965,[172] Hiltibran 1967).[140] (See the discussion of Pesticides, pp. 182–186, in Section III.) On man-made lakes, reservoirs and ponds the potential for invasion by undesirable aquatic plants may be lessened by employing naturalistic methods which limit the available habitat and requirements of particular species. It is difficult to predict what biotic form will replace the species eliminated. Boyd (1971b)[126] states that in some Florida lakes, herbicide applications have upset the balance between rooted aquatics and phytoplankton, resulting in nuisance phytoplankton blooms that were sometimes more objectionable than the original situation.

Control of aquatic vascular plants can be a positive factor in fisheries management (Leonard and Cain 1961);[146] but when control projects are contemplated in multi-purpose waters, consideration should be given to existing interdependencies between man and the aquatic community. For example: what biomass of aquatic vascular plants is necessary to support waterfowl; what biomass will permit boating; what is a tolerable condition for swimming; must the shoreline be clear of plants for wading; will shore erosion increase if the shoreline vegetation is removed? The interference of aquatic vascular plant communities in human activities should be controlled with methods that stop short of attempted plant eradication.

Recommendation

The complex interrelationships among aquatic vascular plants, associated biota, water quality, and the activities of humans call for case-by-case evaluation in assessing the need for management programs. If management is undertaken, study of its potential impacts on the aquatic ecosystem and on various water uses should precede its implementation.

INTRODUCTION OF SPECIES

Extent and Types of Introductions

Purposeful or accidental introductions of foreign aquatic organisms or transplantations of organisms from one drainage system to another can profoundly influence the aesthetic appeal and the recreational or commercial potential of affected waterbodies. The introduction of a single species may alter an entire aquatic ecosystem (Lachner et al. 1970).[188] An example of extreme alteration occurred with the invasion of the Great Lakes by the sea lamprey (*Petromyzon marinus*) (Moffett 1957,[190] Smith 1964[197]). Introduced and transplanted species account for about half of the fish fauna of Connecticut (Whitworth et al. 1968),[199] California (Shapovalov et al. 1959),[195] Arizona, and Utah (Miller 1961).[189] The nature of the original aquatic fauna is obscured in many cases, and some indigenous species have been adversely affected through predation, competition, hybridization, or alteration of habitat by the introduced species. Exotics that have established reproducing populations in the United States (exclusive of the Hawaiian Islands) include 25 species of fish (Lachner et al. 1970),[188] more than 50 species of land and aquatic mollusks (Abbott 1950),[178] and over 20 species of aquatic vascular plants (Hotchkiss 1967)[185] in addition to aquatic rodents, reptiles, amphibians, insects, and crustaceans.

Growths of native aquatic vascular plants and a variety of exotic species commonly interfere with recreation and fishing activities (see p. 25) and a variety of other water uses including industrial and agricultural use (Holm et al. 1969,[184] Sculthorpe 1967).[194] Water hyacinth (*Eichhornia crassipes*) caused loss of almost $43 million through combined deleterious effects in Florida, Alabama, Mississippi, and Louisiana in 1956 (Wunderlich 1962).[200] Penfound and Earle (1948)[192] estimated that the annual loss caused by water hyacinth in Louisiana before the growths were brought under control averaged $5 million and in some years reached $15 million. Water chestnut (*Trapa natans*) produced beds covering 10,000 acres within ten years of its introduction near Washington, D.C. (Rawls 1964).[193] The beds blocked navigation and provided breeding sites for mosquitoes, and their hard spined seed cases on the shorelines and bottom were a serious nuisance to swimmers, waders, and people walking the shores. Eurasian milfoil (*Myriophyllum spicatum*) infested 100,000 acres in Chesapeake Bay. The plants blocked navigation, prevented recreational boating and swimming, interfered with seafood harvest, increased siltation, and encouraged mosquitoes (Cronin 1967).[182]

Invertebrate introductions include the Asian clam (*Corbicula manilensis*), a serious pest in the clogging of industrial and municipal raw water intake systems and irrigation canals (Sinclair 1971),[196] and an oriental oyster drill (*Tritonalia japonica*) considered the most destructive drill in the Puget Sound area (Korringa 1952).[187]

Some Results of Introductions

Some introductions of exotics, e.g., brown trout (*Salmo trutta*), and some transplants, e.g., striped bass (*Morone saxatilis*) from the Atlantic to the Pacific and coho salmon (*Oncorhynchus kisutch*) from the Pacific to the Great Lakes, have been spectacularly successful in providing sport and commercial fishing opportunities. Benefits of introductions and transplantations of many species in a variety of aquatic situations are discussed by several authors in *A Century of Fisheries in North America* (Benson 1970).[179]

The success of other introductions has been questionable or controversial. In the case of carp (*Cyprinus carpis*), the introduction actually decreased aesthetic values because of the increased turbidity caused by the habits of the carp. The increased turbidity in turn decreased the biological productivity of the waterbody. The presence of carp has lowered the sportfishing potential of many waterbodies because of a variety of ecological interactions. The grass carp or white amur (*Ctenopharyngodon idella*), a recent importation, has been reported from several major river systems including the Mississippi as far north as Illinois (Lopinot *personal communication* 1972).[201] Pelzman (1971),[191] in recommending against introducing grass carp into California, concluded that their impact on established game fish would be detrimental and that they might become more troublesome than the common carp. This view was expressed earlier by Lachner et al. (1970)[188] in considering the impact of establishment of the species in major river systems. The walking catfish (*Clarias batrachus*), accidentally released from outdoor holding ponds of aquarium fish dealers in southern Florida, quickly established reproducing populations in a variety of habitats (Idyll 1969).[186] Natural ponds have produced up to 3,000 pounds per acre of this species and there is no current American market for its flesh. This aggressive and omniverous species apparently reduces the entire freshwater community to walking catfish (Lachner et al. 1970).[188]

Introductions by Official Agencies

The objectives of introductions of new species by agencies include pond culture; aquatic plant control; insect control; forage; predation; and improvement of sport and commercial fishing. Boating, swimming, and sport and commercial fin and shellfishing are influenced by water quality and the biotic community. Lachner et al. (1970),[188] after reviewing the history of exotic fish releases, concluded that most official releases satisfy certain social wishes but have not served effective biological purposes, and that some may result in great biological damage. The guidelines of Craighead and Dasmann (1966)[181] on introduction of exotic big game species offer an excellent parallel to the considerations that should precede the introduction of aquatic organisms. Such guidelines call for (a) the establishment of the need and determination of the predicted ecological, recreational, and economic impact; (b) studies of the proposed release area to determine that it is suitable, that a niche is vacant, and that indigenous populations will not be reduced or displaced; (c) life history studies of the organism to determine possible disease interrelationships, hybridization potential, and the availability of control technology; and (d) experiments conducted under controlled conditions that indicate how to prevent escape of the organism.

The California Fish and Game Commission (Burns 1972)[180] investigated introducing the pancora (*Aegla laevis laevis*), a small freshwater crab, into streams as a food for trout to increase natural trout production and sport fishing potential. The plan was ultimately rejected, but the on-site studies in Chile and the experimental work in California illustrate the breadth of consideration necessary before any informed decision can be reached. Problems associated with introductions of aquatic animals were the subject of two recent symposia (Stroud 1969;[198] Department of Lands and Forests, Ottawa 1968[183]). Persons contemplating introductions are referred for guidelines to the Committee on Exotic Fishes and Other Aquatic Organisms of The American Fisheries Society. This committee has representation from the American Society of Ichthyologists and Herpetologists and is currently expanding the scope of its membership to include other disciplines.

Recommendations

Introduction or transplantation of aquatic organisms are factors that can affect aesthetics, boating, swimming, sport and commercial fin and shellfishing, and a variety of other water uses. Thorough investigations of an organism's potential to alter water quality, affect biological relationships, or interfere with other water uses should precede any planned introductions or transplantations.

The deliberate introduction of non-indigenous aquatic vascular plants, particularly in the warmer temperature or tropical regions, is cautioned against because of the high potential of such plants for impairing recreational and aesthetic values. Aquaculturists and others should use care to prevent the accidental release of foreign species for the same reasons.

WATER QUALITY FOR GENERAL RECREATION, BATHING, AND SWIMMING

Historically, public health officials have been concerned about the role of sewage-contaminated bathing water in the transmission of infectious disease. In 1921, the Committee on Bathing Places, Sanitary Engineering Section, American Public Health Association, conducted a study "to determine the extent and prevalence of infections which may be conveyed by means of swimming pools and other bathing places" (Simons et al. 1922).[226] The results of the study, though inconclusive, suggested that contaminated bathing water may transmit infectious agents to bathers. The Committee attached special importance to the data they collected on epidemics of conjunctivitis and other skin diseases, middle ear infections, tonsillitis, pharyngitis, and nasal sinus infections caused by contaminated bathing waters. However, the 1935 Report of the Committee (now designated as the Joint Committee on Bathing Places of the Public Health Engineering Section of the American Public Health Association and the Conference of State Sanitary Engineers) included the following statement: "The summary of the replies in the 1921 report when considered in the light of known epidemiological evidence, leaves this committee unconvinced that bathing places are a *major* public health problem even though bathing place sanitation, because of the health considerations involved, should be under careful surveillance of the public health authorities, and proper sanitary control of bathing places should be exercised" (Yearbook of APHA 1936).[202]

The suggested standards for design, equipment, and operation of bathing places that were part of the 1935 report included a section entitled "Relative Classification of Bathing Areas Recommended" (Yearbook of APHA 1936).[202] This section reads, in part, as follows:

> In passing on waters of outdoor bathing places, three aides are available: (1) the results of chemical analyses of the water; (2) the results of bacteriological analysis of the water; and (3) information obtained by a sanitary survey of sources of pollution, flow currents, etc.—It is not considered practicable or desirable to recommend any absolute standards of safety for the waters of outdoor bathing places on any of the three above bases.

In 1939 (Yearbook of APHA 1940)[203] and again in 1955 (Yearbook of APHA 1957),[204] the Joint Committee surveyed all state health departments for additional information on reported cases of illness attributable to bathing places, but these surveys uncovered little definite information. Contaminated bathing waters were suspected in cases of sleeping sickness, sinus infections, intestinal upsets, eye inflammation, "swimmers itch", ear infections, and leptospirosis.

Several outbreaks of human leptospirosis, which is primarily an infection of rats and dogs, have been associated with recreational waters contaminated by the urine of infected animals (Diesch and McCulloch 1966).[210] One source of infection to man is wading or swimming in waters contaminated by cattle wastes (Williams et al. 1956,[231] Hovens et al. 1941[216]). Leptospirosis is prevalent among "wet crop" agricultural workers, employees of abattoirs, handlers of livestock, and those who swim in stock-watering ponds. The organism is not ingested but enters the body through breaks in the skin and through intact mucous membrane, particularly the conjunctiva.

The most recent reports on disease associated with swimming suggest that a free-living, benign, soil and water amoeba of the *Naegleria* group (*Acanthamoeba*) may be a primary pathogen of animals and man. Central nervous system amoebiasis is usually considered a complication of amoebic dysentery due to *E. histolytical*; however, recent evidence proves that *Naegleria gruberi* causes fulmenting meningoencephalitis (Callicot 1968,[208] Butt 1966,[207] Fowler and Carter 1965,[212] Patras and Andujar 1966[224]). The amoeba may penetrate the mucous membrane. Free-living amoebae and their cysts are rather ubiquitous in their distribution on soil and in natural waters; and identifiable disabilities from free-living amoebae, similar to the situation with leptospirosis, occur so rarely as a result of recreational swimming in the United States that both may be considered epidemiological curiosities (Cerva 1971).[209]

In 1953, the Committee on Bathing Beach Contamination of the Public Health Laboratory Service of England and Wales began a five-year study of the risk to health from bathing in sewage-polluted sea water and considered "the practicability of laying down bacteriological standards for bathing beaches or grading them according to degree of pollution to which they are exposed" (Moore 1959).[222] This committee concluded in 1959 that "bathing in sewage-polluted sea water carries only a negligible risk to health, even on beaches that are aesthetically very unsatisfactory."

The consensus among persons who have studied the relationship between bathing water quality and bathers' illness appears to be that scientific proof of a direct relationship is lacking, yet there is evidence to suggest that [illegible] relationship exists. Some experts contend that outbreaks of illness among bathers have not been studied thoroughly with modern epidemiologic techniques, and that if such occurrences were to be studied vigorously, specific knowledge about the relationship of bathing water quality to infectious disease would be established. In some studies where bathing water was apparently implicated in the transmission of disease agents, the water quality was relatively poor, yet no attempts were made to define the specific relationship.

Water quality requirements for recreational purposes may be divided into two categories: (1) general requirements that pertain to all recreational waters, and (2) special requirements, usually more restrictive, for selected recreational use of water.

GENERAL REQUIREMENTS FOR ALL RECREATIONAL WATERS

Aesthetic Considerations

As has been stressed earlier in this Section (See Applying Recommendations, p. 10), all waters should be aesthetically pleasing, but the great variety of locales makes it impossible to apply recommendations without considering the particular contexts. Color of swamp waters would hardly be acceptable for clear mountain streams. Specific recommendations should reflect adequate study of local background quality and should consider fully the inherent variability so that the designated values will be meaningful. Therefore, specific local recommendations might better encompass ranges, or a daily average further defined by a sampling period, and possibly an absolute maximum or minimum as appropriate. The best technical thought should be given to establishment of such values rather than dependence on administrative or judicial decision.

Recommendation

All recreational surface waters will be aesthetically pleasing if they meet the recommendations presented in the discussion of Water Quality for Preserving Aesthetic Values in this Section, p. 12.

Microbiological Considerations

The hazard posed by pathogenic microorganisms in recreational water not intended for bathing and swimming is obviously less than it would be if the waters were [illegible] those purposes, but it is not possible to state [illegible] degree. Although there is a paucity of [illegible] data on illnesses caused by bathing [illegible] swimming, there appear to be no data that [illegible] relationship of the quality of [illegible] not intended for bathing and swimming [illegible] health of persons enjoying such waters. Criteria concerning the presence of microorganisms in water for general recreation purposes are not known.

Conclusion

No specific recommendation concerning the microbiological qualities of general recreational waters is presented. In most cases of gross microbiological pollution of surface waters, there will be concomitant foreign substance of such magnitude as to cause the water to be aesthetically unacceptable.

Chemical Considerations

The human body is capable of tolerating greater concentrations of most chemicals upon occasional contact with or ingestion of small quantities of water than are most forms of aquatic life. Therefore, specific recommendations for the chemical characteristics of all recreational waters are not made since such recommendations probably would be superseded by recommendations for the support of various forms of desirable aquatic life. (See Sections III and IV: Freshwater and Marine Aquatic Life and Wildlife.)

Recommendations

No specific recommendation concerning the chemical characteristics of general recreational waters is presented. However, the following general recommendations are applicable:

- **recreational waters that contain chemicals in such concentrations as to be toxic to man if small quantities are ingested should not be used for recreation;**
- **recreational waters that contain chemicals in such concentrations as to be irritating to the skin or mucous membranes of the human body upon brief immersion are undesirable.**

SPECIAL REQUIREMENTS FOR BATHING AND SWIMMING WATERS

Since bathing and swimming involve intimate human contact with water, special water quality requirements apply to designated bathing and swimming areas. These requirements are based on microbiological considerations, temperature and pH, and clarity and chemical characteristics. They are more precise than the requirements for general recreational waters. If a body of water cannot meet these specialized requirements, it should not be designated a bathing and swimming area but may be designated for a recreational use that does not involve planned immersion of the body.

Microbiological Considerations

All recreational waters should be sufficiently free of pathogenic bacteria so as not to pose hazards to health through infections, but this is a particularly important requirement for planned bathing and swimming areas. Many bodies of water receive untreated or inadequately treated human and animal wastes that are a potential focus of human infection.

There have been several attempts to determine the specific hazard to health from swimming in sewage-contaminated water. Three related studies have been conducted in this country, demonstrating that an appreciably higher overall illness incidence may be expected among swimmers than among nonswimmers, regardless of the quality of the bathing water (Smith et al. 1951,[229] Smith and Woolsey 1952,[227] 1961[228]). More than one half of the illnesses reported were of the eye, ear, nose, and throat type; gastrointestinal disturbances comprised up to one-fifth; skin irritations and other illnesses made up the balance.

Specific correlation between incidence of illness and bathing in waters of a particular bacterial quality was observed in two of the studies. A statistically significant increase in the incidence of illness was observed among swimmers who used a Lake Michigan beach on three selected days of poorest water quality when the mean total coliform content was 2,300 per 100 ml. However, only the data concerning these three days could be used in the analysis and differences in illness were not noted in comparison with a control beach over the total season (Smith et al. 1951).[229] The second instance of positive correlation was observed in an Ohio River study where it was shown that, despite the relatively low incidence of gastrointestinal disturbances, swimming in river water having a median coliform density of 2,700 per 100 ml appears to have caused a statistically significant increase in illnesses among swimmers (Smith and Woolsey 1952).[227] No relationship between illness and water quality was observed in the third study conducted at salt water beaches on Long Island Sound (Smith and Woolsey 1961).[228]

A study in England suggested that sea water carries only a negligible risk to health even on beaches that were aesthetically unsatisfactory (Moore 1959).[222] The minimal risk attending such bathing is probably associated with chance contact with fecal material that may have come from infected persons.

Neither the English nor the United States salt water beach studies indicated a causal or associated relationship between water quality and disease among swimmers and bathers. While the two United States fresh water studies suggested some presumptive relationship, the findings were not definitive enough to establish specific values for microbiological water quality characteristics.

Tests using fecal coliform bacteria are more indicative of the possible presence of enteric pathogenic microorganisms from man or other warm-blooded animals than the coliform group of organisms. The data for total coliform levels of the Ohio River Study were reevaluated to determine comparable levels of fecal coliform bacteria (Geldreich 1966).[213] This reevaluation suggested that a density of 400 fecal coliform organisms per 100 ml was the approximate equivalent of 2,700 total coliform organisms per 100 ml. Using these data as a basis, a geometric mean of 200 fecal coliform organisms per 100 ml has been recommended previously as a limiting value that under normal circumstances should not be exceeded in water intended for bathing and swimming (U.S. Department of the Interior, FWPCA 1968).[230]

There may be some merit to the fecal coliform index as an adjunct in determining the acceptability of water intended for bathing and swimming, but caution should be exercised in using it. Current epidemiological data are not materially more refined or definitive than those that were available in 1935. The principal value of a fecal coliform index is as an indicator of possible fecal contamination from man or other warm-blooded animals. A study of the occurrence of *Salmonella* organisms in natural waters showed that when the fecal coliform level was less than 200 organisms per 100 ml, this group of pathogenic bacteria was isolated less frequently (Geldreich 1970).[214] *Salmonella* organisms were isolated in 28 per cent of the samples with a fecal coliform density less than the 200 value, but they were isolated in more than 85 per cent of the samples that exceeded the index value of 200 fecal coliform per 100 ml, and in more than 98 per cent of the samples with a fecal coliform density greater than 2,000 organisms per 100 ml.

In evaluating microbiological indicators of recreational water quality, it should be remembered that many of the diseases that seem to be causally related to swimming and bathing in polluted water are not enteric diseases or are not caused by enteric organisms. Hence, the presence of fecal coliform bacteria or of *Salmonella sp.* in recreational waters is less meaningful than in drinking water. Indicators other than coliform or fecal coliform have been suggested from time to time as being more appropriate for evaluating bathing water quality. This includes the staphylococci (Favero et al. 1964),[211] streptococci and other enterococci (Litsky et al. 1953).[218] Recently *Pseudomonas aeruginosa*, a common organism implicated in ear infection, has been isolated from natural swimming waters (Hoadley 1968)[215] and may prove to be an indicator of health hazards in swimming water. Unfortunately, to date, none of the alternative microbiological indicators have been supported by epidemiological evidence.

When used to supplement other evaluative measurements, the fecal coliform index may be of value in determining the sanitary quality of recreational water intended for bathing and swimming. The index is a measure of the "sanitary cleanliness" of the water and may denote the possible presence of untreated or inadequately treated human wastes. But it is an index that should be used only in conjunction with other evaluative parameters of water quality such as sanitary surveys, other biological indices of pollution, and chemical analyses of water. To use the fecal coliform index as the sole measure of "sanitary cleanliness," it would be necessary to know the maximum "acceptable" concentration of organisms; but there is no agreed-upon value that divides "acceptability" from "unacceptability."* Thus, as a measure of "sanitary cleanliness," an increasing value in the fecal coliform index denotes simply a decrease in the level of cleanliness of the water.

Conclusion

No specific recommendation is made concerning the presence or concentrations of microorganisms in bathing water because of the paucity of valid epidemiological data.

Temperature Characteristics

The temperature of natural waters is an important factor governing the character and extent of the recreational activities, primarily in the warm months of the year. Persons engaging in winter water recreation such as ice skating, duck hunting, and fishing do so with the knowledge that whole body immersion must be avoided. Accidental immersion in water at or near freezing temperatures is dangerous because the median lethal immersion time is less than 30 minutes for children and most adults (Molnar 1946).[220] Faddists swim in water that is near the freezing temperature, but their immersion time is short, and they have been conditioned for the exposure. As a result of training, fat insulation, and increased body heat production, some exceptional athletic individuals (Korean pearl divers and swimmers of the English channel) can withstand prolonged immersion for as long as 17 hours in water at 16 C (61 F), whereas children and some adults might not survive beyond two hours (Kreider 1964).[217]

From one individual to another, there is considerable variation in the rates of body cooling and the incidence of survival in cold water. The variability is a function of body size, fat content, prior acclimatization, ability to exercise, and overall physical fitness. The ratio of body mass to surface area is greater in large, heavy individuals, and their mass changes with temperature more slowly than that of a small child (Kreider 1964).[217]

TABLE I-3—Life Expectancy in Water

(Expected duration in hours for adults wearing life vests and immersed in waters of varying temperature)

Duration hours	Temperature of the water								
	32	41	50	59	68	78	86	95	104 F°
	0	5	10	15	20	25	30	35	40 C°
0.5	M	M	S	S	S	S	S	S	M
1.0	L	M	M	S	S	S	S	S	L
2.0	L	L	M	M	S	S	S	S	L
3.0	L	L	L	M	S	S	S	S	L
4.0	L	L	L	L	M	S	S	S	L

L = Lethal, 100 per cent expectancy of death.
M = Marginal, 50 per cent expectancy of unconsciousness, probably drowning.
S = Safe, 100 per cent survival.
Adapted from tables by Pan American Airways and others.

With the exception of water temperatures affected by thermal springs, ocean currents such as the Gulf Stream, and man-made heat, the temperature of natural water is the result of air temperature, solar radiation, evaporation, and wind movement. Many natural waters are undesirably cold for complete body immersion even during the summer period. These include coastal waters subjected to cold currents such as the Labrador Current on the northeastern coastline or the California Current in the Pacific Ocean (Meyers et al. 1969).[219] In addition, some deep lakes and upwelling springs, and streams and lakes fed from melting snow may have summer surface temperatures too cold for prolonged swimming for children.

The most comfortable temperature range for instructional and general recreational swimming where the metabolic rate of heat production is not high—i.e., about 250 kilo calories/hr (1000 BTUs/hr)—appears to be about 29–30 C (84–86 F). In sprint swimming when metabolic rates exceed 500 kilo calories/hr (2,000 BTUs/hr), swimmers can perform comfortably in water temperatures in the range of 20–27 C (68–80 F) (Bullard and Rapp 1970).[206]

The safe upper limit of water temperature for recreational immersion varies from individual to individual and seems to depend on psychological rather than physiological considerations. Unlike cold water, the mass/surface area ratio in warm water favors the child. Physiologically, neither adult nor child would experience thermal stress under modest metabolic heat production as long as the water temperature was lower than the normal skin temperature of 33 C (91 F) (Newburgh 1949).[223] The rate at which heat is conducted from the immersed human body is so rapid that thermal balance for a body at rest in water can only be attained if the water temperature is about 34 C (92 F) (Beckman 1963).[205] The survival of an individual submerged in water at a temperature above 34–35 C (93–95 F), depends on his tolerance to the elevation of his internal temperature, and there is a real risk of injury with prolonged exposure (Table I-3). Water ranging in temperature from 26–30 C (78–86 F) is comfortable to most swimmers throughout prolonged periods of moderate physical exertion (Bullard and Rapp 1970).[206] Although data are limited, natural surface waters do not often exceed skin temperature, but water at 32 C (90 F) is not unusual for rivers and estuaries (Public Works 1967).[225]

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* If an arbitrary value for the fecal coliform index is desired, consideration may be given to a density value expressed as a geometric mean of a series of samples collected during periods of normal seasonal flow. A maximum value of 1,000 fecal coliform per 100 ml could be considered.

Recommendation

In recreational waters used for bathing and swimming, the thermal characteristics should not cause an appreciable increase or decrease in the deep body temperature of bathers and swimmers. One hour of continuous immersion in waters colder than 15 C (59 F) may cause the death of some swimmers and will be extremely stressful to all swimmers who are not garbed in underwater protective cold-clothing. Scientific evidence suggests that prolonged immersion in water warmer than 34–35 C (93–94 F) is hazardous. The degree of hazard varies with water temperature, immersion time, and metabolic rate of the swimmer.

pH Characteristics

Some chemicals affect the pH of water. Many saline, naturally alkaline, or acidic fresh waters may cause eye irritation because the pH of the water is unfavorable. Therefore, special requirements concerning the pH of recreational waters may be more restrictive than those established for public water supplies.

The lacrimal fluid of the human eye has a normal pH of approximately 7.4 and a high buffering capacity due primarily to the presence of complex organic buffering agents. As is true of many organic buffering agents, those of the lacrimal fluid are able to maintain the pH within a narrow range until their buffering capacity is exhausted. When the lacrimal fluid, through exhaustion of its buffering capacity, is unable to adjust the immediate contact layer of another fluid to a pH of 7.4, eye irritation results. A deviation of no more than 0.1 unit from the normal pH of the eye may result in discomfort, and appreciable deviation will cause severe pain (Mood 1968).[221]

Ideally, the pH of swimming water should be approximately the same as that of the lacrimal fluid, i.e., 7.4. However, since the lacrimal fluid has a high buffering capacity, a range of pH values from 6.5 to 8.3 can be tolerated under average conditions. If the water is relatively free of dissolved solids and has a very low buffering capacity, pH values from 5.0 to 9.0 may be acceptable to most swimmers.

Conclusion

For most bathing and swimming waters, eye irritation is minimized and recreational enjoyment enhanced by maintaining the pH within the range of 6.5 and 8.3 except for those waters with a low buffer capacity where a range of pH between 5.0 and 9.0 may be tolerated.

Clarity Considerations

It is important that water at bathing and swimming areas be clear enough for users to estimate depth, to see subsurface hazards easily and clearly, and to detect the submerged bodies of swimmers or divers who may be in difficulty. Aside from the safety factor, clear water fosters enjoyment of the aquatic environment. The clearer the water, the more desirable the swimming area.

The natural turbidity of some bathing and swimming waters is often so high that visibility through the water is dangerously limited. If such areas are in conformance with all other requirements, they may be used for bathing and swimming, provided that subsurface hazards are removed and the depth of the water is clearly indicated by signs that are easily readable.

Conclusion

Safety and enhancement of aesthetic enjoyment is fostered when the clarity of the water in designated bathing and swimming areas allows the detection of subsurface hazards or submerged bodies. Where such clarity is not attainable, clearly readable depth indicators are desirable.

Chemical Considerations

It is impossible to enumerate in specific terms all the specialized requirements that pertain to the chemical quality of bathing and swimming waters. In general, these requirements may be quantified by analyzing the conditions stipulated by two kinds of human exposure, i.e., ingestion and contact. A bather involuntarily swallows only a small amount of water while swimming, although precise data on this are lacking.

Recommendation

Prolonged whole body immersion in the water is the principal activity that influences the required chemical characteristics of recreational waters for bathing and swimming.

The chemical characteristics of bathing and swimming waters should be such that water is nontoxic and nonirritating to the skin and the mucous membranes of the human body. (See also the Recommendations on p. 30.)

WATER QUALITY CONSIDERATIONS FOR SPECIALIZED RECREATION

The recreational enjoyment of water involves many activities other than water contact sports. Some of these, such as boating, may have an adverse effect on the quality of water and require berthing and launching facilities that in themselves may degrade the aesthetic enjoyment of the water environment. Others, such as fishing, waterfowl hunting, and shellfish harvesting, depend upon the quality of water being suitable for the species of wildlife involved. Because they are water-related and either require or are limited by specific water-quality constituents for their continuance, these specialized types of recreation are given individual attention.

BOATING

Boating is a water-based recreational activity that requires aesthetically pleasing water for its full enjoyment. Boats also make a contribution to the aesthetic and recreational activity scene as the sailboat or canoe glides about the water surface or the water skier performs. Boating activity of all types has an element of scale with larger and faster boats associated with larger waterbodies. Many of the problems associated with boating are essentially violations of scale.

Boating activities also have an impact on water quality. The magnitude of the impact is illustrated by recent estimates that there are more than 12 million pleasure boats in the United States (Outboard Boating Club 1971).[235] More than 8 million of these are equipped with engines, and 300,000 have sanitary facilities without pollution control devices. Because of the large number of boats in use, many bodies of water are now experiencing problems that adversely affect other water uses, such as public water supply, support of aquatic life, and other types of water-based recreation.

The detrimental effect of boating on water quality comes from three principal sources: waste disposal systems, engine exhaust, and refuse thrown overboard. Discharges from waste disposal systems on boats are individually a small contribution to contamination and may not be reflected in water-quality sampling, but they represent a potential health hazard and an aesthetic nuisance that must be controlled in or near designated swimming areas. Pathogens in human waste are probably the most important contaminant in the discharges, because of their potential effect on human health (see discussions on Special Requirements for Bathing and Swimming Waters, p. 30, and Shellfish, p. 36). Biochemical oxygen demand (BOD) and suspended solids (SS) are also involved in the discharges, but the quantities are not likely to have any measurable effect on overall water quality. In view of this, it would appear that primary emphasis should be on the control of bacteria from sanitary systems.

The exhaust of internal combustion engines and the unburned fuel of the combustion cycle affect aesthetic enjoyment and may impart undesirable taste and odors to water supplies and off-flavors to aquatic life. Crankcase exhaust from the two-cycle engine can discharge as much as 40 per cent of the fuel to the water in an unburned state, while 10 to 20 per cent is common (Muratori 1968).[233] One study showed that the use of 2.2–3.5 gal/acre-foot (using an oil:fuel mixture of 1:17) will cause some indication of fish flesh tainting, and about 6 gal/acre-foot result in severe tainting (English et al. 1963).[232] (For further discussions of the effects of oil on environments, see Sections III and IV on Freshwater and Marine Aquatic Life and Wildlife.)

The amount of lead emitted into the water from an outboard motor burning leaded gasoline (0.7 grams of lead per liter) appears to be related to the size of the motor and the speed of operation. A 10-hp engine operated at one-half to three-fourths throttle was shown to emit into the water 0.229 grams of lead per liter of fuel consumed, whereas a 5.6-hp engine operated at full throttle emitted 0.121 grams per liter (English et al. 1963).[232]

With respect to interference with other beneficial uses, it has been reported that a large municipal water works is experiencing difficulties with oil on the clarification basins. The oil occurs subsequent to periods of extensive weekend boating activity during the recreational season (Orsanco Quality Monitor 1969).[234] Moreover, bottles, cans, plastics, and miscellaneous solid wastes commonly deface waters where boaters are numerous, thereby degrading the environment aesthetically.

Waste discharge including sanitary, litter, sullage, or bilge from any water craft substantially reduces the water quality of harbors and other congested areas. The practice is aesthetically undesirable and may constitute a health hazard. When engine emissions from boats spread an oily film on water or interfere with beneficial uses, as in lowering the value of fish and other edible aquatic organisms by imparting objectionable taste and odor to their flesh, restrictions should be devised to limit engine use or reduce the emissions.

Floating or submerged objects affect boating safety, and stray electrical currents increase corrosion as do corrosive substances or low pH values. Growth of hull-fouling organisms is enhanced by the discharge of high-nutrient-bearing wastewaters. These conditions represent either a hazard to boating or an economic loss to the boat operator.

Conclusion

Water that meets the general recommendations for aesthetic purposes is acceptable for boating. (see Water Quality for Preserving Aesthetic Values, pp. 11–12.)

Boats and the impact of boating on water quality are factors affecting the recreational and aesthetic aspects of water use and should be considered as such.

AQUATIC LIFE AND WILDLIFE

Fish, waterfowl, and other water-dependent wildlife are an integral part of water-based recreation activities and related aesthetic values. Wildlife enhances the aesthetic quality of aquatic situations by adding animation and a fascinating array of life forms to an otherwise largely static scene. Observation of these life forms, whether for photographic, educational nature study, or purely recreational purposes, is an aesthetically enriching experience. The economic importance and popularity of recreation involving the harvest of fish, shellfish, waterfowl, and water-dependent furbearers have been discussed earlier. Water-quality characteristics recommended for the well-being of aquatic life and associated wildlife are discussed in detail in Sections III and IV on Freshwater and Marine Aquatic Life and Wildlife.

Maintenance of Habitat

Pressures placed on the aquatic environment by the increasing human population are of major concern. They often lead at least to disruption and occasionally to destruction of related life-support systems of desired species. Examples of this are the complete elimination of aquatic ecosystems by the filling of marshes or shallow waters for commercial, residential, or industrial developments, or the sometimes chronic, sometimes partial, and sometimes total destruction of aquatic communities by society's wastes. Effects of cultural encroachment are often insidious rather than spectacular. Aesthetic values are gradually reduced, as is recruitment of water-associated wildlife populations.

Maintenance of life-support systems for aquatic life and water-related wildlife requires adequately oxygenated water, virtual freedom from damaging materials and toxicants, and the preservation of a general habitat for routine activities, plus the critical habitat necessary for reproduction, nursery areas, food production, and protection from predators. Each species has its specific life-support requirements that, if not adequately met, lead to depauperate populations or complete species elimination. The life-support systems essential to the survival of desired aquatic life and wildlife are required for man to enjoy the full scope of water-related recreational and aesthetic benefits.

Man is often in direct competition for a given habitat with many species of aquatic life and wildlife. In some areas, the use of specific waters for recreation based on aquatic life and wildlife may be undesirable for a number of reasons, including potential conflicts with other recreational activities. Limitations on the use of surface water capable of providing recreational wildlife observation, hunting, and fishing under practical management should not be imposed by unsuitable water quality.

Variety of Aquatic Life

Natural surface waters support a variety of aquatic life, and each species is of interest or importance to man for various reasons. While water-based recreation often evokes thoughts of fishing, there are a number of other important recreational activities, such as skin diving, shell and insect collecting, and photography, that also benefit from the complex interrelationships that produce fish. A variety of aquatic life is intrinsic to our aesthetic enjoyment of the environment. Urban waterbodies may be the only local sites where residents can still conveniently observe and contemplate a complete web of life, from primary producers through predators.

Reduction in the variety of aquatic life has long been widely used as an indication of water-quality degradation. The degree of reduction in species diversity often indicates the intensity of pollution because, as a general rule, as pollution increases, fewer species can tolerate the environment. Determining the extent of reduction can be accomplished by studying the entire ecosystem; but the phenomenon is also reflected in the community structure of subcomponents, e.g., bottom animals, plankton, attached algae, or fish. Keup et al. (1967)[236] compiled excerpts of early studies of this type. Mackenthun (1969)[237] presented numerous case studies dealing with different types of pollutants, and Wilhm and Dorris (1968)[238] have reviewed recent efforts to express diversity indices mathematically.

While most water quality recommendations in Sections III and IV on Freshwater and Marine Aquatic Life and Wildlife are designed for specific and known hazards, it is impossible to make recommendations which will protect all organisms from all hazards, including manipulation of the physical environment. In similar habitats and under similar environmental conditions, a reduction in variety of aquatic life (species diversity) can be symptomatic of an ecosystem's declining health and signal deterioration of recreational or other beneficial uses. In addition to maintenance of aquatic community structures, special protective consideration should be given sport, commercial, and endangered species of aquatic life and wildlife.

Recommendations

To maintain and protect aesthetic values and recreational activities associated with aquatic life and wildlife, it is recommended that the water quality recommendations in the Freshwater and Marine Aquatic Life and Wildlife reports (Sections III and IV) be applied.

Since changes in species diversity are often associated with changes in water quality and signal probable changes in recreational and aesthetic values, it is recommended that changes in species diversity be employed as indications that corrective action may be necessary. (See Section III on Freshwater Aquatic Life and Wildlife, and Appendix II–B on Community Structure and Diversity Indices.)

SHELLFISH

Shellfish* are a renewable, manageable natural resource of considerable economic importance, and the water quality essential to their protection in estuarine growing areas is discussed by the panel on Marine Aquatic Life and Wildlife (Section IV). However, the impact of shellfish as related to recreational and aesthetic enjoyment is also important, although difficult to estimate in terms of time and money. Furthermore, because contaminated shellfish may be harvested by the public, it is necessary to protect these people and others who may eat the unsafe catch.

Clams and oysters are obtained from intertidal areas, and these marine species have an unusual ability to act as disease vectors and to accumulate hazardous materials from the water. As more people are able to seek them in a sports fishery, the problems of public health related to these animals intensify.

Because the intent here is to protect persons engaged in recreational shellfishing, consideration will be given to numerous factors which affect shellfish and their growing areas. These include bacteriological quality, pesticides, marine biotoxins, trace metals, and radionuclides.

Recreational shellfishing should be limited to waters of quality that allow harvesting for direct marketing. Epidemiological evidence accumulated through 46 years of operation under the federal–state cooperative National Shellfish Sanitation Program (NSSP) demonstrated reasonable safety in taking shellfish from approved growing areas.

The water quality criteria for determining an "approved growing area" are the basis of the standards given in the National Shellfish Sanitation Program Manual of Operations, Part 1, *Sanitation of Shellfish Growing Areas* (PHS Pub No. 33, 1965).[261] The growing area may be designated as "approved" when:

(a) the sanitary survey indicates that pathogenic microorganisms, radionuclides, or toxic wastes do not reach the area in dangerous concentrations; and

(b) potentially dangerous concentrations are verified by laboratory findings whenever the sanitary survey indicates the need.

Bacteriological Quality

Clams and oysters, which are capable of concentrating bacteria and viruses, are among the few animals eaten alive and raw by man. For these reasons, the consumption of raw shellfish harvested from unclean or polluted waters is dangerous. Polluted water, especially that receiving domestic sewage, may contain high numbers of bacteria normally carried in the feces of man and other animals. Although these bacteria may not themselves be harmful, the danger exists that pathogenic bacteria and viruses may also be present (Lumsden et al. 1925,[250] Old and Gill 1946,[257] Mason and McLean 1962,[251] Mosley 1964a,[254] 1964b;[255] Koff et al. 1967).[248] Shellfish are capable of pumping prodigious quantities of water in their feeding and concentrating the suspended bacteria and viruses. The rate of feeding in shellfish is temperature-dependent, with the highest concentrating and feeding rate occurring in warm water above 50 F and almost no feeding occurring when the water temperatures approach 32 F. Therefore, shellfish meat in the winter months will have a lower bacterial concentration than in the summer months (Gibbard et al., 1942).[246] The National Shellfish Sanitation Program determines the bacteriological quality of commercial shellfish harvesting areas in the following manner:

- examinations are conducted in accordance with the recommended procedures of the American Public Health Association for the examination of seawater and shellfish:
- there must be no direct discharges of inadequately treated sewage;
- samples of water for bacteriological examination are collected under those conditions of time and tide which produce maximum concentrations of bacteria:
- the coliform median most probable number (MPN) of the water does not exceed 70 per 100 ml, and not more than 10 per cent of the samples ordinarily exceed an MPN of 230 per 100 ml for a five-tube

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* As used here, the term "shellfish" is limited to clams, oysters, and mussels.

decimal dilution test (or 330 per 100 ml for a three-tube decimal dilution test) in those portions of the area most probably exposed to fecal contamination during the more unfavorable hydrographic and pollution conditions; and

- the reliability of nearby waste treatment plants is considered before areas for direct harvesting are approved.

Recommendation

Recreational harvesting of shellfish should be limited to areas where water quality meets the National Shellfish Sanitation Program Standards for approved growing areas.

Pesticides

Pesticides reach estuarine waters from many sources including sewage and industrial waste discharge, runoff from land used for agriculture and forestry, and chemicals used to control aquatic vegetation and shellfish predators. Once pesticides are in the marine environment, they are rapidly accumulated by shellfish, sometimes to toxic concentrations. Organochlorine compounds are usually the most toxic and frequently have a deleterious effect at concentrations near 0.1 µg/l in the ambient water (Butler 1966b).[240] Lowe (1965)[249] observed that DDT at a concentration of 0.5 µg/l in water was fatal to juvenile blue crabs (*Callinectes sapidus*) in a few days.

The biological magnification of persistent pesticides by mollusks in the marine environment may be very pronounced. Butler (1966a)[239] observed that DDT may be concentrated to a level 25,000 times that found in surrounding sea water within 10 days. In some instances, depending upon water temperature, duration of exposure, and concentration of DDT in the surrounding water, biological magnification may be 70,000 times (Butler 1966b).[240] Some shellfish species, particularly blue mussel (*Mytilus edulis*), appear to have a higher concentration factor than other species (Modin 1969,[253] Foehrenbach 1972).[245]

In 1966, a nationwide surveillance system was initiated by the U.S. Bureau of Commercial Fisheries to monitor permanent mollusk populations and determine the extent of pesticide pollution in North American estuaries. Butler (1969)[241] reported that sampling during the first three years did not indicate any consistent trends in estuarine pesticide pollution. Distinct seasonal and geographical differences in pollution levels were apparent. Pesticides most commonly detected in order of frequency were DDT (including its metabolites), endrin, toxaphene, and mirex. The amounts detected in North American estuaries varied. In Washington, less than 3 per cent of the sampled shellfish were contaminated with DDT. Residues were always less than 0.05 mg/l. On the Atlantic Coast, DDT residues in oysters varied from less than 0.05 mg/l in marine estuaries to less than 0.5 mg/l in others. In a monitoring program for chlorinated hydrocarbon pesticides in estuarine organisms in marine waters of Long Island, New York, Foehrenbach (1972)[245] found that residues of DDT, DDD, DDE, and dieldrin in shellfish were well within the proposed limits of the 6th National Shellfish Sanitation Workshop (1968)[262] (see Table I-4). For most cases, the levels detected were 10- to 20-fold less than the recommendations for DDT and its metabolites, and in many instances concentrations in the shellfish were lower by a factor of 100.

TABLE I-4—Recommended Guidelines for Pesticide Levels in Shellfish

Pesticide	Concentration in shellfish (ppm—drained weight)
Aldrin[a]	0.20
BHC	0.20
Chlordane	0.03
DDT)	
DDE) ANY ONE OR ALL, NOT TO EXCEED	1.50
DDD)	
Dieldrin[a]	0.20
Endrin[a]	0.20
Heptachlor[a]	0.20
Heptachlor Epoxide[a]	0.20
Lindane	0.20
Methoxychlor	0.20
2,4-D	0.50

[a] It is recommended that if the combined values obtained for Aldrin, Dieldrin, Endrin, Heptachlor, and Heptachlor Epoxide exceed 0.20 ppm, such values be considered as "alert" levels which indicate the need for increased sampling until results indicate the levels are receding. It is further recommended that when the combined values for the above five pesticides reach the 0.25 ppm level, the areas be closed until it can be demonstrated that the levels are receding.

U.S. Department of Health, Education and Welfare, Public Health Service 1968.[212]

Although pesticide levels in many estuaries in the United States are low, the marked ability of shellfish to concentrate pesticides indicates that the levels approached in waters may be considered significant in certain isolated instances (Environmental Protection Agency 1971).[244]

Recommendation

Concentrations of pesticides in fresh and marine waters that provide an adequate level of protection to shellfish are recommended in the Freshwater and Marine Aquatic Life and Wildlife Reports, Sections III and IV. Levels that protect the human consumer of shellfish should be based on pesticide concentrations in the edible portion of the shellfish. Recommended human health guidelines for pesticide concentrations in shellfish have been suggested by the 6th National Shellfish Sanitation Workshop (1968)[262], Table I-4. They are recommended here as interim guidelines.

Marine Biotoxins

Paralytic poisoning due to the ingestion of toxic shellfish, while not a major public health problem, is a cause of concern to health officials because of its extreme toxicity, and because there is no known antidote. Up to 1962, more than 957 cases of paralytic shellfish poisoning are known to have occurred, resulting in at least 222 deaths in the United States (Halstead 1965).[24]

Paralytic shellfish poison is a non-protein, acid-stable, alkali-labile biotoxin nearly 10,000 times as lethal as sodium cyanide. The original source of the poison is a species of unicellular marine dinoflagellates, genus *Gonyaulax*. *Gonyaulax cantenella*, perhaps the best known of the toxic dinoflagellates, is found on the Pacific Coast. *Gonyaulax tamarensis* is the causative organism of paralytic shellfish poison on the Atlantic Coast of Canada and the northern United States. Other dinoflagellate species have been identified in outbreaks of paralytic shellfish poisoning outside the United States (Halstead 1965).[247]

Mollusks and other seashore animals may become poisonous if they consume toxic planktonic algae. Mussels and clams are the principal species of edible mollusks that reach dangerous levels of toxicity. Although oysters can also become toxic, their apparent uptake of toxin is usually lower; and they are usually reared in areas free of toxin (Dupuy and Sparks 1968).[248]

The level of toxicity of shellfish is proportional to the number and poison content of *Gonyaulax* ingested. When large numbers of *Gonyaulax* are present in the water, shellfish toxicity may rise rapidly to dangerous levels (Prakash and Medcof 1962).[258] The extent of algal growth depends on the combination of nutrients, salinity, sunlight, and temperature. Massive blooms of algae are most likely to occur in the warm summer months. In the absence of toxic algae, the poison that had been stored in the shellfish is eliminated by a purging action over a period of time (Sommer and Meyer 1937).[260]

Although *Gonyaulax* only blooms in the warmer months, shellfish are not necessarily free from toxin during the rest of the year, as there is great variation in the rates of uptake and elimination of the poison among the various species of mollusks. It is possible for certain species to remain toxic for a long period of time. Butter clams, for example, store the toxin for a considerable length of time, especially under cold climatic conditions (Chambers and Magnusson 1950).[242]

Cooking by boiling, steaming, or pan frying does not remove the danger of intoxication, although it does reduce the original poison content of the raw meat to some extent. Pan frying seems to be more effective than other cooking methods in reducing toxicity probably because higher temperatures are involved. If the water in which shellfish have been boiled is discarded, most of the toxin will be removed (McFarren et al. 1965).[252]

A chemical method for the quantitative determination of the poison has been devised, but the most generally used laboratory technique for determining the toxicity of shellfish is a bioassay using mice. The toxin extracted from shellfish is injected into test mice and the length of time elapsing from injection of the mice to the time of their death can be correlated with the amount of poison the shellfish contain. The quantity of paralytic shellfish poison producing death is measured in mouse units.

Recommendation

Since there is no analytical measurement for the biotoxin in water, shellfish should not be harvested from any areas even if "approved" where analysis indicates a *Gonyaulax* shellfish toxin poison content of 80 micrograms or higher, or where a *Ciguateria*-like toxin reaches 20 mouse units per 100 grams of the edible portions of raw shellfish meat.

Trace Metals

The hazard to humans of consuming shellfish containing toxic trace metals has been dramatized by outbreaks of Minimata in Japan. Pringle et al. (1968)[259] noted that the capacity of shellfish to concentrate *in vivo* some metals to levels many hundred times greater than those in the environment means that mollusks exposed to pollution may contain quantities sufficient to produce toxicities in the human consumer.

Recommendation

Concentrations of metals in fresh and marine waters that provide an adequate level of protection to shellfish are recommended in the Freshwater and Marine Aquatic Life and Wildlife Sections, III and IV. Recommendations to protect the human consumer of shellfish should be based on trace-metal content of the edible portions of the shellfish, but necessary data to support such recommendations are not currently available.

Radionuclides

Radioactive wastes entering water present a potential hazard to humans who consume shellfish growing in such water. Even though radioactive material may be discharged into shellfish growing waters at levels not exceeding the applicable standards, it is possible that accumulation of radionuclides in the aquatic food chain may make the organisms used as food unsafe. The radionuclides Zn^{65} and P^{32} (National Academy of Sciences 1957)[256] are known to be concentrated in shellfish by five orders of magnitude (10^5). Therefore, consideration must be given to radioactive fallout or discharges of wastes from nuclear reactors and industry into shellfish growing areas. For further discussion of this subject see Section IV, Marine Aquatic Life and Wildlife, p. 270.

WATER QUALITY CONSIDERATIONS FOR WATERS OF SPECIAL VALUE

WILD AND SCENIC RIVERS

There are still numerous watersheds in the United States that are remote from population centers. Almost inaccessible and apparently free from man's developmental influences, these watersheds are conducive to mental as well as physical relaxation in the naturalness of their surroundings. To assure the preservation of such natural beauty, the Wild and Scenic Rivers Act of 1968 established in part a national system of wild and scenic rivers (U.S. Congress 1968).[269] Eight rivers designated in the Act in whole or in part constituted the original components of the system:

1. Clearwater, Middle Fork, Idaho
2. Eleven Point, Missouri
3. Feather, California
4. Rio Grande, New Mexico
5. Rogue, Oregon
6. Saint Croix, Minnesota and Wisconsin
7. Salmon, Middle Fork, Idaho
8. Wolf, Wisconsin

All or portions of 27 other rivers were mentioned specifically in the Act as being worthy of inclusion in the system if studies to be conducted by several federal agencies showed their inclusion to be feasible. Certainly there are many more rivers in the nation worthy of preservation by state and local agencies (U.S. Department of the Interior, Bureau of Outdoor Recreation, 1970).[270] In Kentucky alone, it was found that 500 streams and watersheds, near urban areas, would serve purposes of outdoor recreation in natural environments (Dearinger 1968).[264]

Characteristically, such wild river areas are: (a) accessible to man in only limited degrees; (b) enjoyed by relatively few people who actually go to the site; (c) visited by scout troops or other small groups rather than by lone individuals; and (d) productive of primarily intangible, aesthetic benefits of real value though difficult to quantify (U.S. Outdoor Recreation Resources Review Commission 1962,[271] Sonnen et al. 1970[267]).

The quality of natural streams is generally good, primarily because man's activities leading to waste discharges are minimal or nonexistent in the area.* However, fecal coliform concentrations in some natural waters have been found to be quite high following surface runoff (Betson and Buckingham *unpublished report* 1970;[273] Kunkle and Meiman 1967[266]), indicating the possible presence of disease-causing organisms in these waters. The sources of fecal coliforms in natural waters are wild and domestic animals and birds, as well as human beings who occasionally visit the area. Barton (1969)[263] has also reported that natural areas may contribute significant loads of nitrogen, phosphorus, and other nutrients to the streams that drain them. These chemicals can lead to algal blooms and other naturally occurring but aesthetically unpleasant problems. Barton (1969)[263] also points out the paradox that a significant contributor to pollution of natural waters is the human being who comes to enjoy the uniquely unpolluted environment. In addition to water-quality degradation, man also contributes over one pound per day of solid wastes or refuse in campgrounds and wilderness areas, a problem with which the Forest Service and other agencies must now cope (Spooner 1971).[268]

This discussion has concentrated on Wild and Scenic rivers. However, similar consideration should be given to the recognition and preservation of other wild stretches of ocean shoreline, marshes, and unspoiled islands in fresh and salt waters.

WATER BODIES IN URBAN AREAS

Many large water bodies are located near or in urban and metropolitan areas. These waters include major coastal estuaries and bays, portions of the Great Lakes, and the largest inland rivers. Characteristically, these waters serve a multiplicity of uses and are an economic advantage to the region and to the nation as a whole. In addition to providing water supplies, they have pronounced effects on local weather and make possible valuable aesthetic and recreational pleasures, ranging from simple viewing to fishing and boating.

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* Some of the least mineralized natural waters are those in high mountain areas fed by rainfall or snowmelt running across stable rock formations. One such stream on the eastern slope of the Rocky Mountains has been found to have total dissolved solids concentrations often below 50 mg/l, coliform organism concentrations of 0 to 300/ml, and turbidities of less than 1 unit (Kunkle and Meiman 1967).[266]

Large urban waterways, because of their location in densely populated areas, are heavily used commercially and are also in great demand for recreational and aesthetic purposes. Consequently, although swimming and other contact activities cannot always be provided in all such waters, quality levels supportive of these activities should be encouraged.

Water flow in the urban stream tends to be variable and subject to higher and more frequent flood flows than under "natural conditions," because storm water runoff from buildings and hard-surfaced areas is so complete and rapid. The impaired quality of the water may be due to storm water runoff, upstream soil erosion, or sewage discharges and low base flow. Whitman (1968)[272] surveyed the sources of pollution in the urban streams in Baltimore and Washington and reported that sewer malfunctions, many of which might be eliminated, were the largest causes of poor water quality. In large metropolitan areas with either separate or combined sewer systems, pollution of the urban waterway can be expected during heavy rainstorms when the streams may contain coliform concentrations in the millions per 100 ml. In addition, these flood waters flow with treacherous swiftness and are filled with mud and debris.

Small urban streams are even more numerous. Although these may have only intermittent flow, they have the capacity to provide considerable opportunities for a variety of water-related recreation activities. Unfortunately, these in-city streams are more often eyesores than they are community treasures. Trash, litter, and rubble are dumped along their banks, vegetation is removed, channels are straightened and concrete stream beds are constructed or even roofed over completely to form covered sewers.

This abuse and destruction of a potential economic and social resource need not occur. The urban stream can be made the focal point of a recreation-related complex. The needs of the cities are many, and not the least of them is the creation of a visually attractive urban environment in which the role of water is crucial.

The reclamation of downtown sections of the San Antonio River in the commercial heart of San Antonio, Texas, is perhaps the best known and most encouraging example of the scenic and cultural potential of America's urban streams (Gunn et al. 1971).[265] From a modest beginning with WPA labor in the mid-1930's, the restoration of about a one-mile portion of the river threading its way through the central business district has resulted in the creation of the Paseo Del Rio, or River Walk. Depressed below the level of adjacent streets, heavily landscaped with native and tropical vegetation, the river is bordered with pleasant promenades along which diners relax in outdoor cafes. Fountains and waterfalls add to the visual attractiveness, and open barges carry groups of tourists or water-borne diners to historic buildings, restaurants, clubs, and a River Theater. More popular with both local residents and tourists each year, the River Walk has proved to be a significant social and economic development, attracting commercial enterprises to a previously blighted and unattractive area. The River Walk is widely visited and studied as a prototype for urban river reclamation, and it demonstrates that urban rivers can serve as the environmental skeleton on which an entire community amenity of major proportions can be built.

OTHER WATERS OF SPECIAL VALUE

Between the remote and seldom used waters of America at one extreme and the urban waterways at the other are many unique water recreation spots that are visited and enjoyed by large numbers of tourists each year. Among these are Old Faithful, Crater Lake, The Everglades, the Colorado River-Grand Canyon National Park, and Lake Tahoe. These ecologically or geologically unique waters are normally maintained in very nearly their natural conditions, but access to them is freer and their monetary value is greater than that of the wild rivers. To many, however, their aesthetic value will always be greater than their monetary value. It is obviously impossible to establish nationally applicable quality recommendations for such waters. (It would be ludicrous, for example, to expect Old Faithful to be as cool as Crater Lake, or The Everglades as clear as Lake Tahoe.) Nonetheless, responsible agencies should establish recommendations for each of these waters that will protect and preserve their unique values.

Municipal raw water supply reservoirs are often a potential source of recreation and aesthetic enjoyment. Periodic review of the recreational restrictions to protect water quality in such reservoirs could result in provision of additional recreational and aesthetic opportunities. (See also the general Introduction, p. 3–4 regarding preservation of aquatic sites of scientific value.)

Conclusions

To preserve or enhance recreational and aesthetic values:

- **water quality supportive of general recreation is adequate to provide for the intended uses of wild and scenic rivers;**
- **water quality supportive of general recreation is adequate to protect or enhance uses of urban streams, provided that economics, flow conditions, and safety considerations make these activities feasible;**
- **special criteria are necessary to protect the nation's unique recreational waters with regard to their particular physical, chemical, or biological properties.**

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INTRODUCTION

Modern water management techniques and a wide variety of available water treatment processes make possible the use of raw water of almost any quality to produce an acceptable public water supply. For this reason it is both possible and desirable to consider water management alternatives and treatment procedures in making recommendations on the quality of raw water needed for public supplies. Furthermore, these recommendations must be consistent with the effort and money it is reasonable to expect an individual, company, or municipality to expend to produce a potable water supply. Defining a reasonable effort including treatment processes involves consideration of present water quality, the degree of improvement in raw water that is attainable within the bounds of natural controls on water quality, and the help that can be expected from society in cleaning up its waters. In evaluating the basis for the recommendations in this Section, the Panel has left water management alternatives open wherever possible, but it has made certain arbitrary assumptions about the treatment process.

The federal Drinking Water Standards for treated water for public supply (U.S. Department of Health, Education and Welfare, Public Health Service 1962, hereafter referred to as PHS 1962[6])* are under review and revision, but the final standards were not available to the Panel on Public Water Supplies at the time of publication of this Report. The Panel did, however, have access to the data, references, and rationale being considered in the revision of Drinking Water Standards, and these have had a major influence on recommendations in this report.

THE DEFINED TREATMENT PROCESS

Surface water supplies characteristically contain suspended sediment in varying amounts and are subject to bacterial and viral contamination. Therefore, it is assumed that the following defined treatment, and no more, will be given raw surface water in a properly operated plant prior to human consumption.

1. coagulation (less than about 50 milligrams per liter (mg/l) alum, ferric sulfate, or copperas with alkali or acid addition as necessary but without coagulant aids or activated carbon);
2. sedimentation (6 hours or less);
3. rapid sand filtration (three gallons per square foot per minute or more);
4. disinfection with chlorine (without consideration to concentration or form of chlorine residual).

The panel recognizes that on the one hand some raw surface waters will meet federal Drinking Water Standards with no treatment other than disinfection, and that on the other hand almost any water, including sea water and grossly polluted fresh water, can be made potable for a price by available treatment processes already developed. However, the defined treatment outlined above is considered reasonable in view of both the existing and generally attainable quality of raw surface waters, and the protection made imperative by the current practice of using streams to transport and degrade wastes. Assumption of the defined treatment process throughout this Section is not meant to deny the availability, need, or practicality of other water treatment processes.

Unlike surface waters, ground waters characteristically contain little or no suspended sediment and are largely free of and easily protected from bacterial and viral contamination. (See *Ground Water Characteristics* below for significant exceptions.) Therefore, no defined treatment is assumed for raw ground water designated for use as a public supply, although here again this does not deny the availability, need, or practicality of treatment. Ground waters should meet current federal Drinking Water Standards in regard to bacteriological characteristics and content of toxic substances, thus permitting an acceptable public water supply to be produced with no treatment, providing natural water quality is adequate in other respects. The recommendations in this section based on considerations other than bacterial content and toxicity apply to ground waters as well as surface waters unless otherwise specified.

WATER QUALITY RECOMMENDATIONS

The Panel has defined water quality recommendations as those limits of characteristics and concentrations of sub-

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*** Citations are listed at the end of the Section. They can be located alphabetically within subtopics or by their superior numbers which run consecutively across subtopics for the entire Section.**

stances in raw waters that will allow the production of a safe, clear, potable, aesthetically pleasing and acceptable public water supply after treatment. In making these recommendations, the Panel recognized that most of the surface water treatment plants providing water for domestic use in the United States are relatively small, do not have sophisticated technical controls, and are operated by individuals whose training in modern methods varies widely. The recommendations assume the use of the treatment process defined above but no more.

Regional variations in natural water quality make it necessary to apply understanding and discretion when evaluating raw water quality in terms of the recommendations. Wherever water zoned for public supply fail to meet the recommendations in all respects, the recommendations can be considered the minimum goal toward which to work in upgrading water quality. In some instances the natural presence of certain constituents in raw water sources may make the attainment of recommended levels impractical or even impossible. When such constituents affect human health, the water cannot be used for public supply unless the constituent can be brought to Drinking Water Standards levels through a specially designed treatment process prior to distribution to consumers. Where health is not a factor, the natural level of the constituent prior to man-made additions can be considered a reasonable target toward which to work, although determination of "natural quality" may require considerable effort, expense, and time.

The recommendations in this report should by no means be construed as latitude to add substances to waters where the existing quality is superior to that called for in the recommendations. Degradation of raw water sources of quality higher than that specified should be minimized in order to preserve operational safety factors and economics of treatment.

The Panel considered factors of safety for each of the toxic substances discussed, but numerical factors of safety have been employed only where data are available on the known no-effect level or the minimum effect level of the substances on humans. These factors were selected on the basis of the degree of hazard and the fraction of daily intake of each substance that can reasonably be assigned to water.

The recommendations should be regarded as guides in the control of health hazards and not as fine lines between safe and dangerous concentrations. The amount and length of time by which values in the recommendations may be exceeded without injury to health depends upon the nature of the contaminant, whether high concentrations even for short periods produce acute poisoning, whether the effects are cumulative, how frequently high concentrations occur, and how long they last. All these factors must be considered in deciding whether a hazardous situation exists.

Although some of the toxic substances considered are known to be associated with suspended solids in raw surface waters and might thus be removed to some extent by the defined treatment process, the degree of removal of the various dissolved toxic substances is not generally known; and even if known, it could not be assured under present treatment practices. Therefore, in the interest of safety, it has usually been assumed here that there is no removal of toxic substances as a result of the defined treatment process.

Substances not evaluated in this Section are not necessarily innocuous in public water supply sources. It would be impractical to prepare a compendium of all toxic, deleterious, or otherwise unwelcome agents, both organic and inorganic, that may enter a surface water supply. In specific locations it may become necessary to consider substances not included in this section, particularly where local pollution suggests that a substance may have an effect on the beneficial use of water for public supplies.

In summary: *the recommendations in this Section for raw water quality for public supplies are intended to assure that the water will be potable—for surface water, with the defined treatment process; for ground water, with no treatment. For waters zoned for public supply but not meeting the recommendations in all respects, the recommendations can be considered a minimum target toward which efforts at upgrading the quality should be directed. In some instances the natural quality of raw water may make meeting certain recommendations impractical or even impossible. For constituents for which this is the case, and where health is not a factor, the natural quality of the water can be considered a reasonable target toward which to work, although determination of "natural quality" may require considerable effort, expense, and time. Wherever water quality is found superior to that described in the recommendations, efforts should be made to minimize its degradation.*

SAMPLING AND MONITORING

The importance of establishing an effective sampling and monitoring program and the difficulties involved cannot be overemphasized. A representative sample of the water entering the raw water intake should be obtained. Multiple sampling, chronologically and spatially, may be necessary for an adequate characterization of the raw water body, particularly for constituents associated with suspended solids (Great Britain Department of the Environment 1971;[3] Brown et al. 1970;[2] Rainwater and Thatcher 1960[4]). Monitoring plans should take into account the results of sanitary surveys (U.S. Department of Health, Education, and Welfare 1969;[7] American Public Health Association, American Water Works Association, and Water Pollution Control Federation 1971[1] hereafter referred to as Standard Methods 1971[5]) and the possibility of two types of water quality hazards: (1) the chronic hazard where constituent concentrations are near the limit of acceptability much of the time, and (2) the periodic hazard caused by upstream release of wastes or accidental spills of hazardous substances into the stream. Samples for the determination of dissolved constituents only should be passed through a noncontaminating filter at time of collection.

ANALYTICAL METHODS

The recommendations are based on the use of analytical methods for raw water analysis as described in Standard Methods (1971).[5] Other procedures of similar scientific acceptability are continuously evolving but whatever the analytical procedure used, the panel assumes that it will conform to the statistical concepts of precision, accuracy, and reporting style discussed in the introduction to Standard Methods (1971).[5] Analytical results should indicate whether they apply to a filtered or unfiltered sample.

GROUND WATER CHARACTERISTICS

Development of water quality recommendations for ground water must provide for the significant differences between surface water and ground water. Ground water is generally not confined in a discrete channel. Its quality can be measured in detail only with difficulty and at great expense. A thorough knowledge of the hydrologic characteristics of the ground water body can be obtained only after extensive study. Movement of ground water can be extremely slow so that contamination occurring in one part of an aquifer may not become evident at a point of withdrawal for several, tens, hundreds, or even thousands of years.

Wastes mix differently with ground waters than they do with surface waters. Where allowance for a mixing zone in the immediate vicinity of a waste outfall can be provided for in surface water standards under the assumption that mixing is complete within a short distance downstream, dispersion of waste in a ground water body may not be complete for many years. At the same time, the long retention time will facilitate bacterial or chemical reactions with aquifer components that result in removal or decomposition of a pollutant to the point where it no longer degrades the aquifer. Because these reactions are imperfectly known and cannot be predicted at the present time, it is necessary to monitor the movement of waste in a ground water body from the point of introduction outward. Bodies of ground water cannot be monitored adequately by sampling at the point of use.

Inadvertent or careless contamination of fresh ground water bodies is occurring today from the leaching of accumulated salts from irrigation, animal feed lots, road salt, agricultural fertilizers, dumps, and landfills, or from leakage of sewer lines in sandy soil, septic tank effluents, petroleum product pipelines, and chemical waste lagoons. Another source of contamination is the upward movement of saline water in improperly plugged wells and drill holes, or as the result of excessive withdrawal of ground water. Deep-well injection causes intentional introduction of wastes into saline ground water bodies.

Because of their common use as private water supplies in rural areas, all geologically unconfined (water-table) aquifers could be placed in a classification comparable to that for raw surface waters used for public water supplies. Even though not all waters in these aquifers are suitable for use without treatment, such classification could be used to prohibit introduction of wastes into them. This in turn would restrict the use of landfills and other surface disposal practices. Limited use of the unsaturated zone for disposal of wastes would still be acceptable, provided that decomposition of organic wastes and sorption of pollutants in the zone of aeration were essentially complete before the drain water reached the water table. Bodies of artesian ground water in present use as public and private supplies could be similarly classified wherever their natural source of recharge was sufficient to sustain the current yield and quality.

Disposal of wastes in either of the above types of aquifers could be expressly forbidden on the basis of their classification as public water supplies. Furthermore, before disposal of wastes to the soil or bedrock adjacent to aquifers used or usable for public supply were permitted, it could be required that a geologic reconnaissance be made to determine possible effects on ground water quality.

Water quality recommendations for raw ground waters to be used for public water supplies are more restrictive than water quality recommendations for raw surface water source because of the assumption that no treatment will be given to the ground waters. The distinction between surface and ground waters is therefore necessary for proper application of the recommendations. In certain cases this distinction is not easily made. For example, collector wells in shallow river valley alluvium, wells tapping cavernous limestone, and certain other types of shallow wells may intercept water only a short distance away, or after only a brief period of travel, from the point at which it was surface water. Springs used as raw water sources present a similar problem. Choice of the appropriate water quality recommendations to apply to such raw water sources should be based on the individual situation.

WATER MANAGEMENT CONSIDERATIONS

The purpose of establishing water quality recommendations and, subsequently, establishing water quality standards is to protect the nation's waters from degradation and to provide a basis for improvement of their quality. These actions should not preclude the use of good water management practices. For example, it may be possible to supplement streamflow with ground water pumped from wells, or to replace ground water removed from an aquifer with surface water through artificial recharge. These other sources of water may be of lower quality than the water originally present, but it should remain a management choice whether this lower quality is preferable to no water at all. In arid parts of the nation, water management practices of this sort have been applied for many years to partially offset the effects of "mining" of ground water (i.e., its withdrawal faster than it can be recharged naturally).

Furthermore, it is possible, by merely removing ground water from the aquifer, to degrade the quality of that remaining—by inducing recharge from a surface or ground water body of lesser quality. It does not seem reasonable to forbid the use of the high-quality water that is there because of this potential degradation. Of what value is it if it cannot be used?

It would appear, then, that "degradation by choice" might be an alternative under certain conditions and within certain limits. This type of degradation is not comparable to that resulting from disposal of wastes in the water body. It is simply the price exacted for using the water. In the case of mining without artificial recharge, the philosophy involved is the same as that applied to the mining of other nonrenewable resources such as metal ores or fossil fuels. Because considerations of recreation and aesthetics and the maintenance of fish and wildlife are generally not involved in this kind of management situation, it is reasonable that water quality standards should provide for the mining and artificial recharge of bodies of ground water zoned for public supply. As in any water management program, it would be necessary to understand the hydrologic system and to monitor changes induced in the system by management activities.

Preservation of water management choices can be protected by water use classification. Classification of surface waters has not been based solely on the fact that those waters are being used for public supply at the present time. Presumably it has been based on the decision that the body of water in question should be usable for public supply with no more than the routine forms of water treatment, whether or not it is presently in use for that purpose. Conversely, failure to zone a body of water for public supply would not necessarily preclude its use for that purpose. Selective zoning could thus be used to assure desirable water management practices.

ALKALINITY

Alkalinity is a measure of the capacity of a water to neutralize acids. Anions of weak acids such as bicarbonate, carbonate, hydroxide, sulfide, bisulfide, silicate, and phosphate may contribute to alkalinity. The species composition of alkalinity is a function of pH, mineral composition, temperature, and ionic strength.

The predominant chemical system present in natural waters is the carbonate equilibria in which carbonate and bicarbonate ions and carbonic acid are in equilibrium (Standards Methods 1971).[8] The bicarbonate ion is usually more prevalent. A water may have a low alkalinity but a relatively high pH value or vice versa, so alkalinity alone may not be of major importance as a measure of water quality.

The alkalinity of natural waters may have a wide range. An alkalinity below 30 to 50 mg/l, as $CaCO_3$, may be too low to react with hydrolyzable coagulants, such as iron or aluminum salts, and still provide adequate residual alkalinity to produce a water that is not excessively corrosive. Alkalinities below 25 mg/l, as $CaCO_3$, may also lead to corrosive waters when only chlorination is practiced, since there would be inadequate buffer capacity to prevent the pH from dropping appreciably (Weber and Stumm 1963).[9] Low alkalinity waters may be difficult to stabilize by calcium carbonate saturation which would otherwise prevent corrosion of the metallic parts of the system.

High alkalinity waters may have a distinctly unpleasant taste. Alkalinities of natural waters rarely exceed 400 to 500 mg/l (as $CaCO_3$).

Conclusion

No recommendation can be made, because the desirable alkalinity for any water is associated with other constituents such as pH and hardness. For treatment control, however, it is desirable that there be no sudden variations in the alkalinity.

AMMONIA

Ammonia may be a natural constituent of certain ground waters. In surface waters its concentration is normally 0.1 mg/l or less as nitrogen. Higher levels are usually indicative of sewage or industrial contamination (McKee and Wolf 1963).[30]

Ammonia consumes dissolved oxygen as a result of its biochemical oxidation to nitrite and nitrate. Reliance on the biochemical oxygen demand (BOD) test (Standard Methods 1971[33]) for measuring the efficiency of sewage treatment and the quality of effluents has focused attention principally on the oxygen requirements of carbonaceous matter. Ammonia is therefore a common constituent of treated sewage, and much of the burden of satisfying the nitrogenous oxygen demand has, in general, been shifted from the sewage treatment plant to the receiving water (Sawyer and Bradney 1946,[32] Ludzack and Ettinger 1962,[29] Johnson and Schroepfer 1964,[24] Barth et al. 1966,[12] Courchaine 1968,[18] Barth and Dean 1970,[11]; Holden 1970,[22] Barth 1971,[10] Great Britain Department of the Environment 1971,[21] Mt. Pleasant and Schlickenrieder 1971[31]).

Ammonia is sometimes corrosive to copper and copper alloys (LaQue and Copson 1963,[26] Butler and Ison 1966[13]); it is also a potential algal and microbial nutrient in water distribution systems (Larson 1939,[27] Ingram and Mackenthun 1963[23]).

Ammonia has a significant effect on the disinfection of water with chlorine. The reactions of ammonia with chlorine result in the formation of chloramine compounds having markedly less disinfecting efficiency than free chlorine. Ammonia substantially increases the chlorine demand at water treatment plants that practice free-residual chlorination. Approximately 10 parts of chlorine per part of ammonia nitrogen are required to satisfy the ammonia chlorine demand (Butterfield et al. 1943,[16] Butterfield and Wattie 1946,[15] Butterfield 1948,[14] Fair et al. 1948,[19] Kelly and Sanderson 1958,[25] Clarke and Chang 1959,[17] Laubusch 1971[28]). It would therefore be desirable to have as low a level as possible in the raw water.

However, since ammonia is present in ground water and in some surface water supply sources, particularly at cold temperatures, and since it can be removed by the defined treatment process with adequate chlorination, the cost of the treatment is the determining factor. In the previous edition of Water Quality Criteria (U.S. Department of the Interior, Federal Water Pollution Control Administration 1968,[34] hereafter referred to as FWPCA 1968[20]) a permissible level of 0.5 mg/l nitrogen was proposed. This is not a sacrosanct number, but it is considered to be tolerable.

Recommendation

Because ammonia may be indicative of pollution and because of its significant effect on chlorination, it is recommended that ammonia nitrogen in public water supply sources not exceed 0.5 mg/l.

ARSENIC

Arsenic, a metalloid that occurs ubiquitously in nature, can be both acutely and chronically toxic to man. Although no form of arsenic is known to be essential, arsenic has been added in small amounts to animal feed as a growth stimulant. For 1,577 surface water samples collected from 130 sampling points in the United States, 87 samples showed detectable arsenic concentrations of 5 to 336 micrograms per liter (μg/l) with a mean level of 64 μg/l (Kopp 1969).[50]

The chemical forms of arsenic consist of trivalent and pentavalent inorganic and organic compounds. It is not known which forms of arsenic occur in drinking water. Although combinations of all forms are possible, it can be reasonably assumed that the pentavalent inorganic form is the most prevalent. Conditions that favor chemical and biological oxidation promote the shift to the pentavalent species; and conversely, those that favor reduction will shift the equilibrium to the trivalent state.

Arsenic content in drinking water in most United States supplies ranges from a trace to approximately 0.1 mg/l (McCabe et al. 1970).[52] No adverse health effects have been reported from the ingestion of these waters.

Arsenic has been suspected of being carcinogenic (Paris 1820,[55] Sommers and McManus 1953,[60] Buchanan 1962,[38] Frost 1967,[45] Trelles et al. 1970,[61] Borgono and Greiber 1972[36]), but substantial evidence from human experience and animal studies now supports the position that arsenicals are not tumorigenic at levels encountered in the environment (Snegireff and Lombard 1951,[58] Baroni et al. 1963,[35] Boutwell 1963,[37] Hueper and Payne 1963,[47] Pinto and Bennett 1963,[56] Kanisawa and Schroeder 1967,[49] Milner 1969).[53]

Several epidemiological studies in Taiwan (Chen and Wu 1962)[39] have reported a correlation between the increased incidence of hyperkertosis and skin cancer with consumption of water containing more than 0.3 mg/l arsenic. A similar problem has been reported in Argentina (Trelles et al. 1970).[61] Dermatological manifestations of arsenicism were noted in children of Antofagasta, Chile, who used a water supply containing 0.8 mg/l arsenic. A new water supply was provided, and preliminary data showed that arsenic levels in hair decreased (Borgono and Greiber 1972).[36]

Inorganic arsenic is absorbed readily from the gastrointestinal tract, the lungs, and to a lesser extent from the skin and becomes distributed throughout the body tissues and fluids (Sollmann 1957).[59] It is excreted via urine, feces, sweat, and the epithelium of the skin (Dupont et al. 1942,[44] Hunter et al. 1942,[48] Lowry et al. 1942,[51] Ducoff et al. 1948,[43] Crema 1955,[40] Musil and Dejmal 1957).[54] During chronic exposure, arsenic accumulates mainly in bone, muscle, and skin, and to a smaller degree in liver and kidneys. This accumulation can be measured by analysis of hair samples. After cessation of continuous exposure, arsenic excretion may last up to 70 days (DuBois and Geiling 1959).[42]

In man, subacute and chronic arsenic poisoning may be insidious and pernicious. In mild chronic poisoning, the only symptoms present are fatigue and loss of energy. The following symptoms may be observed in more severe intoxication; gastrointestinal catarrh, kidney degeneration, tenency to edema, polyneuritis, liver cirrhosis, bone marrow injury, and exfoliate dermatitis (DiPalma 1965,[41] Goodman and Gilman 1965).[46] It has been claimed that individuals become tolerant to arsenic. However, this apparent effect is probably due to the ingestion of the relatively insoluble, coarse powder, since no true tolerance has been demonstrated (DuBois and Geiling 1959).[42]

The total intake of arsenic from food averages approximately 900 μg/day (Schroeder and Balassa 1966).[57] At a concentration of 0.1 mg/l and an average intake of 2 liters of water per day, the intake from water would not exceed 200 μg/day, or approximately 18 per cent of the total ingested arsenic.

Recommendation

Because of adverse physiological effects on humans and because there is inadequate information on the effectiveness of the defined treatment process in removing arsenic, it is recommended that public water supply sources contain no more than 0.1 mg/l total arsenic.

BACTERIA

Procedures for the detection of disease-causing bacteria, viruses, protozoa, worms, and fungi are complex, time-consuming, and in need of further refinement to increase the levels of sensitivity and selectivity. Therefore, an indirect approach to microbial hazard measurement is required.

Coliform bacteria have been used as indicators of sanitary quality in water since 1880 when *Escherica coli* (*E. coli*) and similar gram negative bacteria were shown to be normal inhabitants of fecal discharges. Although the total coliform group as presently recognized in the Drinking Water Standards includes organisms known to vary in characteristics, the total coliform concept merits consideration as an indicator of sanitary significance, because the organisms are normally present in large numbers in the intestinal tracts of humans and other warm-blooded animals.

Numerous stream pollution surveys over the years have used the total coliform measurement as an index of fecal contamination. However, occasional poor correlations to sanitary significance result from the inclusion of some strains in the total coliform group that have a wide distribution in the environment and are not specific to fecal material. Therefore, interpretation of total coliform data from sewage, polluted water, and unpolluted waters is sometimes difficult For example, *Enterobacter* (*Aerobacter*) *aerogenes* and *Enterobacter cloacae* can be found on various types of vegetation (Thomas and McQuillin 1952,[78] Fraser et al. 1956,[66] Geldreich et al. 1964,[73] Papavassiliou et al. 1967[75]), in soil (Frank and Skinner 1941,[65] Taylor 1951,[77] Randall 1956,[76] Geldreich et al. 1962b[72]), and in water polluted in the past. Also included are plant pathogens (Elrod 1942)[62] and other organisms of uncertain taxonomy whose sanitary significance is questionable. All of these coliform subgroups may be found in sewage and in polluted water.

A more specific bacterial indicator of warm-blooded animal contamination is fecal coliform, defined as those coliform that can ferment lactose at 44.5 C to produce gas in a multiple tube procedure (U.S. Department of Interior, Federal Water Pollution Control Administration 1966[79] hereafter referred to as (FWPCA 1966)[64] or acidity in the membrane filter procedure (M-FC medium: Geldreich et al. 1965).[71] Research showed that 96.14 per cent of the coliform in human feces was positive by this test (Geldreich et al. 1962a).[70] Examination of the excrement from other warm-blooded animals, including livestock, poultry, cats, dogs, and rodents indicates that fecal coliform contribute 93.0 per cent of the total coliform population (FWPCA 1966),[64] Geldreich et al. 1968).[68]

At the present time, the only data available from numerous freshwater stream pollution studies on a correlation of pathogen occurrence with varying levels of fecal coliform are for *Salmonella* (Geldreich 1970,[67] Geldreich and Bordner 1971[69]). These data indicate a sharp increase in the frequency of *Salmonella* detection when fecal coliform densities are above 200 per 100 milliliters (ml). For densities of 1 to 200/100 ml, 41 examinations showed 31.7 per cent positive detection of *Salmonella*. For densities of 201 to 1,000/100 ml, 30 examinations showed 83 per cent positive detection. For densities of 1,000 to 2,000, 88.5 per cent positive detection was found in 17 examinations, and for densities above 2,000, 97.6 per cent positive detection was found in 123 examinations.

The significance is further illustrated by a bacterial quality study at several water plant intakes along the Missouri River. When fecal coliform exceeded 2,000 organisms per 100 ml, *Salmonella*, Poliovirus types 2 and 3, and ECHO virus types 7 and 33 were detected (Environmental Protection Agency 1971).[63] Any occurrence of fecal coliform in water is therefore prime evidence of contamination by wastes of some warm-blooded animals, and as the fecal coliform densities increase, potential health hazards become greater and the challenge to water treatment more demanding.

A study of the bacteriological quality of raw water near six public intakes along the Ohio River showed that of 18 monthly values with maximum total coliform densities in excess of 10,000 organisms per 100 ml, 12 were not paralleled by fecal coliform densities above 2,000 organisms per 100 ml (ORSANCO Water Users Committee 1971).[74] The fecal coliform portion of these total coliform populations ranged from 0.2 to 12 per cent. Data from the Missouri River study showed total coliform densities at water intakes to be frequently in excess of 20,000 organisms per 100 ml with concurrent fecal coliform densities above 2,000 (Environmental Protection Agency 1971).[68] This indicates less coliform aftergrowth, but proportionately more recent fecal pollution.

The major limitation to the total coliform index is the uncertain correlation to the occurrence of pathogenic microorganisms. However, fecal coliform occurrences in water reflect the presence of fecal contamination, which is the most likely source for pathogens.

Total coliform measurements may be used as an alternative to fecal coliform measurements with the realization that such data are subject to a wide range of density fluctuations of doubtful sanitary significance.

A well-operated plant using the defined treatment to process raw surface water meeting the recommendations below can be expected to meet a value of 1 total coliform per 100 ml with proper chlorination practice. When coliform counts in raw surface water approach the recommendations, both pre- and post-chlorination may be required to achieve proper disinfection.

Recommendation

In light of the capabilities of the defined treatment process for raw surface waters and the statistical correlations mentioned, it is recommended that the geometric means of fecal coliform and total coliform densities in raw surface water sources not exceed 2,000/100 ml and 20,000/100 ml, respectively.

BARIUM

Barium (Ba) ingestion can cause serious toxic effects on the heart, blood vessels, and nerves. Barium enters the body primarily through air and water, since essentially no food contains barium in appreciable amounts.

The solubility product of barium sulfate indicates that 1.3 mg/l sulfate ion limits the solubility of barium to 1.0 mg/l. There is some evidence that barium may be adsorbed by oxides or hydroxides of iron and manganese (Ljunggren 1955).[83] For the public water supplies of the 100 largest cities in the United States, the median barium concentration was 0.05 mg/l with a range of 0.01 to 0.058 mg/l. For 1,577 samples of surface waters collected in 130 locations in the United States the barium concentration in 1,568 samples ranged from 2 to 340 µg/l with a mean of 43 µg/l (Kopp 1969).[82]

Barium is recognized as a general muscle stimulant, especially of the heart muscle (Sollmann 1957).[85] The fatal dose for man is considered to be from 0.8 to 0.9 grams(g) as the chloride (550 to 600 mg Ba). Most fatalities have occurred from mistaken use of barium salts incorporated in rat poison. Barium is capable of causing nerve block (Lorente and Feng 1946)[84] and in small or moderate doses produces transient increase in blood pressure by vasoconstriction (Gotsev 1944).[81]

There apparently has been no study made of the amounts of barium that can be tolerated in drinking water, nor any study of the effects of long-term feeding of barium salts from which a standard might be derived. The present barium standard has been developed from the barium-in-air standard, 0.5 mg/cubic meter (m^3) (American Conference of Governmental Industrial Hygienists 1958),[80] based on the retention of innaled barium dusts, and an estimate of the possible adsorption from the intestines (Stokinger and Woodward 1958).[86] This value is 2 mg/l. The air standard provides no indication of the inclusion of a factor of safety. Therefore, it is reasonable to provide a factor of safety of 2 for protection of heterogeneous population.

Recommendation

Because of the adverse physiological effects of barium, and because there are no data on the effectiveness of the defined treatment process on its removal, it is recommended that a limit for barium of 1 mg/l not be exceeded in public water supply sources.

BORON

The previous Report of the Committee on Water Quality Criteria (FWPCA 1968)[87] recommended a permissible limit of 1 mg/l for boron. When a new Drinking Water Standards Technical Review Committee was established in 1971, it determined that the evidence available did not indicate that the suggested limit of 1 mg/l was necessary. More information is required before deciding whether a specific limit is needed for physiological reasons.

Whenever public water supplies are used to irrigate plants, boron concentrations may be of concern because of the element's effect on many plants. For consideration of the possible effect of boron on certain irrigated plants, see Section V on Agricultural Uses of Water (p. 341).

CADMIUM

Cadmium is biologically a nonessential, nonbeneficial element. The possibility of seepage of cadmium into ground water from electroplating plants was reported in 1954 when concentrations ranging from 0.01 to 3.2 mg/l were recorded (Lieber and Welsch 1954).[97] Another source of cadmium contamination in water may be zinc-galvanized iron in which cadmium is a contaminant. For 1,577 surface water samples collected at 130 sampling points in the United States, 40 samples showed detectable concentrations of 1 to 20 μg/l of cadmium with a mean level of 9.5 μg/l. Six samples exceeded 10 μg/l (Kopp 1969).[95]

Cadmium is an element of high toxic potential. Evidence for the serious toxic potential of cadmium is provided by: poisoning from cadmium-contaminated food (Frant and Kleeman 1941)[92] and beverages (Cangelosi 1941);[88] epidemiologic evidence that cadmium may be associated with renal arterial hypertension under certain conditions (Schroeder 1965);[102] epidemiologic association of cadmium with *Itai-itai* disease in Japan (Murata et al. 1970);[99] and long-term oral toxicity studies in animals (Fitzhugh and Meiller 1941,[91] Ginn and Volker 1944,[93] Wilson and DeEds 1950).[104]

Symptoms of violent nausea were reported for 29 school children who had consumed fruit ice sticks containing 13–15 mg/l cadmium (Frant and Kleeman 1941).[92] This would be equivalent to 1.3 to 3.0 mg of cadmium ingested.

It has been stated that the concentration and not the absolute amount determines the acute toxicity of cadmium (Potts et al. 1950).[101] Also, equivalent concentrations of cadmium in water are considered more toxic than concentrations in food because of the effect of components in the food.

The association of cardiovascular disease, particularly hypertension, with ingestion of cadmium remains unsettled. Although conflicting evidence has been reported for man (Schroeder 1965,[102] Morgan 1969)[98] and for animals (Kanisawa and Schroeder 1969,[94] Lener and Bibr 1970[96]), it is notable that hypertension has not been associated with *Itai-itai* disease (Nogawa and Kawano 1969).[100]

In view of the cumulative retention of cadmium by hepatic (liver) and renal (kidney) tissue (Decker et al. 1958,[90] Cotzias et al. 1961,[89] Schroeder and Balassa 1961[103]) and the association of a severe endemic *Itai-itai* disease syndrome with ingestion of as little as 600 μg/day (Yamagata 1970),[105] Drinking Water Standards limit concentrations of cadmium to 10 μg/l so that the maximum daily intake of cadmium from water (assuming a 2 liter daily consumption) will not exceed 20 μg. This is one-third the amount of cadmium derived from food (Schroeder and Balassa 1961).[103] A no-effect level for intake and accumulation of cadmium in man has not been established.

Recommendation

Because of the adverse physiological effects of cadmium, and because there is inadequate information on the effect of the defined treatment process on removal of cadmium, it is recommended that the cadmium concentration in public water supply sources not exceed 0.010 mg/l.

CHLORIDE

Chloride ion in high concentrations, as part of the total dissolved solids in water, can be detected by taste and can lead to consumer rejection of the water supply. In undefined high concentrations it may enhance corrosion of water utility facilities and household appurtenances (American Water Works Association 1971).[106]

For the public water supplies of the 100 largest cities in the United States, the median chloride concentration was 13 mg/l with a range of 0 to 540 mg/l (Durfor and Becker 1964).[107]

The median chloride concentrations detected by taste by a panel of 10 to 20 persons were 182, 160, and 372 mg/l from sodium, calcium, and magnesium salts respectively (Whipple 1907).[110] The median concentration identified by a larger panel of 53 adults was 395 mg/l chloride for sodium chloride (Richter and MacLean 1939).[109] When compared with distilled water for a difference in taste, the median concentration was 61 mg/l. Coffee was affected in taste when brewed with 210 and 222 mg/l chloride from sodium chloride and calcium chloride respectively (Lockhart et al. 1955).[108]

On the basis of taste and because of the wide range of taste perception of humans, and the absence of information on objectionable concentrations, a limit for public water supplies of 250 mg/l chloride appears to be reasonable where sources of better quality water are or can be made available. However, there may be a great difference between a detectable concentration and an objectionable concentration, and acclimatization might be an important factor.

Recommendation

On the basis of taste preferences, not because of toxic considerations, and because the defined treatment process does not remove chlorides, it is recommended that chloride in public water supply sources not exceed 250 mg/l if sources of lower levels are available.

CHROMIUM

Chromium is rarely found in natural waters. It may occur as a contaminant from plating wastes, blowdown from cooling towers, or from circulating water in refrigeration equipment where it is used to control corrosion. It has been found in some foods and in air. Chromium can be detected in most biological systems. This does not prove it essential, although there is reasonable evidence that it does have a biological role (Mertz 1969).[119]

For 1,577 surface water samples collected at 130 sampling points in the United States, 386 samples showed concentrations of 1 to 112 μg/l with a mean concentration of 9.7 μg/l for chromium (Kopp 1969).[116]

The hexavalent state of chromium is toxic to man, produces lung tumors when inhaled (Machle and Gregorius 1948,[117] U.S. Federal Security Agency, Public Health Service 1953[123]), and readily induces skin sensitizations. Trivalent chromium salts show none of the effects of the hexavalent form (Fairhall 1957).[114] The trivalent form is not likely to be present in waters of pH 5 or above because of the very low solubility of the hydrated oxide.

At present, the levels of chromate ion that can be tolerated by man for a lifetime without adverse effects on health are undetermined. It is not known whether cancer will result from ingestion of chromium in any of its valence forms. A family of four individuals is reported to have drunk water for a period of three years with as high as 0.45 mg/l chromium in the hexavalent form without known effects on their health, as determined by a single medical examination (Davids and Lieber 1951).[113]

Levels of 0.45 to 25 mg/l of chromium administered to rats in chromate and chromic ion form in drinking water for one year produced no toxic responses (MacKenzie et al. 1958).[118] However, significant accumulation in the tissues occurred abruptly at concentrations above 5 mg/l. Naumova (1965)[120] demonstrated that 0.033 mg of chromium from potassium bichromate per kilogram (kg) of body weight in dogs enhanced the secretory and motor activity of the intestines. Although there does not appear to be a clearly defined no-effect level, other studies (Conn et al. 1932,[112] Brard 1935,[111] Gross and Heller 1946,[115] Schroeder et al. 1963a,[121] Schroeder et al. 1963b[122]) suggested that a concentration of 0.05 mg/l with an average intake of 2 liters of water per day would avoid hazard to human health.

Recommendation

Because of adverse physiological effects, and because there are insufficient data on the effect of the defined treatment process on the removal of chromium in the chromate form, it is recommended that public water supply sources for drinking water contain no more than 0.05 mg/l total chromium.

COLOR

Color in public water supplies is aesthetically undesirable to the consumer and is economically undesirable to some industries. Colored substances can chelate metal ions, thereby interfering with coagulation (Hall and Packham 1965[130]), and can reduce the capacity of ion exchange resins (Frisch and Kunin 1960).[129] Another serious problem is the ability of colored substances to complex or stabilize iron and manganese and render them more difficult for water treatment processes to remove (Robinson 1963,[135] Shapiro 1964[136]).

Although the soluble colored substances in waters have been studied for over 150 years, there is still no general agreement on their structure. A number of recent studies have indicated that colored substances are a complex mixture of polymeric hydroxy carboxylic acids (Black and Christman 1963a,[125] 1963b,[126] Lamar and Goerlitz 1963,[133] Christman and Ghassemi 1966,[128] Lamar and Goerlitz 1966[134]) with the measurable color being a function of the total organics concentration and the pH (Black and Christman 1963a,[125] Singley et al. 1966[137]).

The removal of color can be accomplished by the defined process when the dosage and the pH are adjusted as functions of the raw water color (Black et al. 1963,[127] American Water Works Association Research Committee on Color Problems 1967).[124] These relationships may not apply to colors resulting from dyes and some other industrial and processing sources that cannot be measured by comparison with the platinum-cobalt standards (Hazen 1892,[131] 1896,[132] Standard Methods 1971[138]). Such colors should not be present in concentrations that cannot be removed by the defined process.

Recommendation

Because color in public water supply sources is aesthetically undesirable and because of the limitations of the defined treatment process, a maximum of 75 platinum-cobalt color units is recommended.

COPPER

Copper is frequently found in surface waters and in some ground waters in low concentrations (less than 1 mg/l). It is an essential and beneficial element in human metabolism, and it is known that a deficiency in copper results in nutritional anemia in infants (Sollmann 1957).[141] Because the normal diet provides only little more than what is required, an additional supplement from water may ensure an adequate intake. Small amounts are generally regarded as nontoxic; but large doses may produce emesis, and prolonged oral administration may result in liver damage.

For 1,577 surface water samples collected at 130 sampling points in the United States, 1,173 showed concentrations of 1 to 280 μg/l with a mean concentration of 15 μg/l (Kopp 1969).[140]

Copper imparts some taste to water, but the detectable range varies from 1 to 5 mg/l (Cohen et al. 1960[139]), depending upon the acuity of individual taste perceptions. Copper in public water supplies enhances corrosion of aluminum in particular and of zinc to a lesser degree. A limit of 0.1 mg/l has been recommended to avoid corrosion of aluminum (Uhlig 1963).[142]

The limit of 1 mg/l copper is based on considerations of taste rather than hazards to health.

Recommendation

To prevent taste problems and because there is little information on the effect of the defined treatment process on the removal of copper, it is recommended that copper in public water supply sources not exceed 1 mg/l.

CYANIDE

Standards for cyanide in water have been published by the World Health Organization in "International Standards for Drinking Water" (1963)[148] and the "European Standards for Drinking Water" (1970).[149] These standards appear to be based on the toxicity of cyanide to fish, not to man. Cyanide in reasonable doses (10 mg or less) is readily converted to thiocyanate in the human body and in this form is much less toxic to man. Usually, lethal toxic effects occur only when the detoxifying mechanism is overwhelmed. The oral toxicity of cyanide for man is shown in the following table.

Proper chlorination with a free chlorine residual under neutral or alkaline conditions will reduce the cyanide level to below the recommended limit. The acute oral toxicity of cyanogen chloride, the chlorination product of hydrogen cyanide, is approximately one-twentieth that of hydrogen cyanide (Spector 1955).[146]

On the basis of the toxic limit calculated from the threshold limit for air (Stokinger and Woodward 1958),[147] and assuming a 2-liter daily consumption of water containing 0.2 mg/l cyanide as a maximum, an appreciable factor of safety would be provided.

TABLE II-1—Oral Toxicity of Cyanide for Man

Dosage	Response	Literature citations
2.9–4.7 mg/day	Noninjurious	Smith 1944[145]
10 mg, single dose	Noninjurious	Bodansky and Levy 1923[143]
19 mg/1 in water	Calculated from the safe threshold limit for air	Stokinger and Woodward 1958[147]
50–60 mg, single dose	Fatal	The Merck Index of Chemicals and Drugs 1968[144]

Recommendation

Because of the toxicity of cyanide, it is recommended that a limit of 0.2 mg/l cyanide not be exceeded in public water supply sources.

DISSOLVED OXYGEN

Dissolved oxygen in raw water sources aids in the elimination of undesirable constituents, particularly iron and manganese, by precipitation of the oxidized form. It also induces the biological oxidation of ammonia to nitrate, and prevents the anaerobic reduction of dissolved sulfate to hydrogen sulfide. More importantly, dissolved oxygen in a raw surface water supply serves as an indicator that excessive quantities of oxygen-demanding wastes are probably not present in the water, although there can be significant exceptions to this. Therefore, it is desirable that oxygen in the water be at or near saturation. On the other hand, oxygen enhances corrosion of treatment facilities, distributing systems, and household appurtenances in many waters.

Oxygen depletion in unmixed bodies of water can result from the presence of natural oxygen-demanding substances as well as from organic pollution. Lakes and reservoirs may contain little or no oxygen, yet may be essentially free of oxygen-demanding wastes. This is because contact with the air is limited to the upper surface, and because thermal stratification in some lakes and reservoirs prevents oxygenation of lower levels directly from the air. Similar conditions also occur in ground waters.

Conclusion

No recommendation is made, because the presence of dissolved oxygen in a raw water supply has both beneficial and detrimental aspects. However, when the waters contain ammonia or iron and manganese in their reduced form, the benefits of the sustained presence of oxygen at or near saturation for a period of time can be greater than the disadvantages.

FLUORIDE

The fluoride ion has potential beneficial effects, but excessive fluoride in drinking water supplies produces objectionable dental fluorosis that increases as a continuum with increasing fluoride concentration above the recommended control limits. In the United States, this is the only harmful effect resulting from fluoride found in drinking water (Dean 1936,[150] Moulton 1942,[158] Heyroth 1952,[155] McClure 1953,[157] Leone et al. 1954,[156] Shaw 1954,[159] U.S. Department of Health, Education, and Welfare, Public Health Service 1959[160]). The fluoride concentrations excessive for a given community depend on climatic conditions because the amount of water (and consequently the amount of fluoride) ingested by children is primarily influenced by air temperature (Galagan 1953,[151] Galagan and Lamson 1953,[152] Galagan and Vermillion 1957,[153] Galagan et al. 1957[154]).

Rapid fluctuations in raw water fluoride ion levels would create objectionable operating problems for treatment plants serving communities that supplement raw water fluoride concentrations. From the point of view of a water pollution control program any value less than that recommended would generally be acceptable at a point of domestic water withdrawal.

Recommendation

Because of adverse physiological effects and because the defined treatment process does nothing to reduce excessive fluoride concentrations, it is recommended that the maximum levels shown in Table II-2 not be exceeded in public water supply sources.

TABLE II-2—Fluoride Recommendation

Annual average of maximum daily air temperatures[a] fahrenheit	Fluoride maximum mg/l
80-91	1.4
72-79	1.6
65-71	1.8
59-64	2.0
55-58	2.2
50-54	2.4

[a] Based on temperature data obtained for a minimum of five years.

FOAMING AGENTS

Many chemical substances occurring either naturally or as components of industrial or domestic waste will cause water to foam when agitated or when air is entrained. The most common foaming agent in use today is the synthetic anionic surfactant, linear alkyl benzene sulfonate (LAS). Branched alkyl benzene sulfonate (ABS) was used prior to 1965 as a base for synthetic detergents. Because of its persistent foaming properties, however, ABS was replaced by LAS. The most objectionable property of surfactants is their foaming capacity which can produce unsightly masses of foam in a stream or at the home tap. The surfactants also tend to disperse normally insoluble or sorbed substances, thus interfering with their removal by coagulation, sedimentation, and filtration.

Although conversion to the more readily biodegradable linear alkyl sulfonates by the detergent industry has decreased the persistence of sulfonates in aerobic waters, measurable concentrations of these substances still can be found in both surface and ground waters. Concentrations of anionic surfactants in water can be determined by means of their reaction with methylene blue dye (Standard Methods 1971).[162] Concentrations of less than 0.5 mg/l, as methylene blue active substances (MBAS), do not cause foaming or present serious interference in the defined treatment process and are well below the inferred limit (700 mg/l) of toxicity to humans based on tests on rats fed diets of LAS (Buehler et al. 1971).[161] It must be recognized that this procedure does not determine the total concentration of foaming agents, merely the concentration of materials that react with methylene blue, most of which are anionic surfactants. Although cationic and nonionic synthetic surfactants do not respond, and not all substances that respond to the methylene blue process cause foaming, the methylene blue test is the best available measure of foaming properties.

Recommendation

To avoid undesirable aesthetic effects and because the defined treatment process does little or nothing to reduce the level of foaming agents, it is recommended that foaming agents determined as methylene blue active substances not exceed 0.5 mg/l in public water supply sources.

HARDNESS

Hardness is defined as the sum of the polyvalent cations expressed as the equivalent quantity of calcium carbonate ($CaCO_3$). The most common such cations are calcium and magnesium. In general, these metal ions in public water supply sources are not cause for concern to health, although there are some indications that they may influence the effect of other metal ions on some organisms (Jones 1938,[168] Cairns, Jr. and Scheier 1958,[164] Mount 1966[170]). Possible beneficial and detrimental effects on health have been postulated but not conclusively demonstrated (Muss 1962,[171] Crawford and Crawford 1967,[166] Crawford et al. 1968,[165] Masironi 1969,[169] Voors 1971[172]). There is considerable variation in the range of hardness acceptable to a given community. Some consumers expect and demand supplies with a total hardness of less than 50 mg/l, expressed as equivalent $CaCO_3$, while others are satisfied with total hardness greater than 200 mg/l. Consumer sensitivity is often related to the hardness to which the public has become accustomed, and acceptance may be tempered by economic considerations.

The requirement for soap and other detergents is directly related to the water hardness (DeBoer and Larson 1961).[167] Of particular importance is the tendency for development of scale deposits when the water is heated. Variations in water hardness may be more objectionable than any given level. Waters with little or no hardness may be corrosive to water utility facilities, depending upon pH, alkalinity, and dissolved oxygen (American Water Works Association 1971).[163] Industrial consumers of public supplies may be particularly sensitive to variations in hardness. A water hardness must relate to the level normal for the supply and exclude hardness additions resulting in significant variations or general increases.

Conclusion

Acceptable levels for hardness are based on consumer preference. No quantitative recommendation for hardness in water can be specified.

IRON

Iron (Fe) is objectionable in public water supplies because of its effect on taste (Riddick et al. 1958,[178] Cohen et al. 1960[175]), staining of plumbing fixtures, spotting of laundered clothes, and accumulation of deposits in distribution systems. Iron occurs in the reduced state (Fe^{++}), frequently in ground waters and less frequently in surface waters, since exposure to oxygen in surface waters results in oxidation, forming hydrated ferric oxide which is much less soluble (American Water Works Association 1971).[173]

Statistical analysis of taste threshold tests with iron in distilled water free of oxygen at pH 5.0 showed that 5 per cent of the observers were able to distinguish between 0.04 mg/l ferrous iron (added as ferrous sulfate) and distilled water containing no iron. At 0.3 mg/l, 20 per cent were able to make the distinction. When colloidal ferric oxide was added, 5 per cent of the observers were able to distinguish between 0.7 mg/l and distilled water. Thus the form of iron is important. The range of sensitivities of the observers was surprising, in that 5 per cent were unable to detect ferrous iron at a concentration of 256 mg/l in distilled water. The taste of iron was variously described as bitter, sweet, astringent, and "iron tasting." (Cohen et al. 1960).[175]

Concentrations of iron less than 0.3 mg/l are generally acceptable in public water supplies as the characteristic red stains and deposits of hydrated ferric oxide do not manifest themselves (Hazen 1895,[176] Mason 1910,[177] Buswell 1928[174]). This is the principal reason for limiting the concentration of soluble iron.

Recommendation

On the basis of user preference and because the defined treatment process can remove oxidized iron but may not remove soluble iron (Fe^{++}), it is recommended that 0.3 mg/l soluble iron not be exceeded in public water supply sources.

LEAD

Lead is well known for its toxicity in both acute and chronic exposures (National Academy of Sciences 1972).[190] In technologically developed countries the widespread use of lead multiplies the risk of exposure of the population to excessive lead levels (Kehoe 1960a).[184] For this reason, constant surveillance of the lead exposure of the general population via food, air, and water is necessary.

Acute lead toxicity is characterized by burning in the mouth, severe thirst, inflammation of the gastrointestinal tract with vomiting and diarrhea. Chronic toxicity produces anorexia, nausea, vomiting, severe abdominal pain, paralysis, mental confusion, visual disturbances, anemia, and convulsions (The Merck Index of Chemicals and Drugs 1960).[189]

For 1,577 surface water samples collected from 130 sampling points in the United States, 11.3 per cent showed detectable concentrations of 0.002 to 0.140 mg/l with a mean of 0.023 mg/l (Kopp 1969).[187] For the 100 largest cities in the United States, the finished waters were found to have a median concentration of 0.0037 mg/l and a maximum of 0.062 mg/l (Durfor and Becker 1964).[182] Of the 969 water supplies in a community water supply study conducted in 1969 (McCabe et al. 1970),[188] the lead concentrations in finished water ready for distribution ranged from 0 to 0.64 mg/l. Fourteen of these supplies on the average exceeded the 0.05 mg/l limit for lead in drinking water (PHS 1962).[191] Of 2,595 samples from distribution systems, 37 exceeded the limit set by the Drinking Water Standards (PHS 1962).[191] When standing in lead pipe overnight, acidic soft water in particular can dissolve appreciable concentrations of lead (Crawford and Morris 1967).[181]

The average daily intake of lead via the diet was 0.3 mg in 1940 and rarely exceeded 0.6 mg (Kehoe et al. 1940a).[186] Data obtained subsequent to 1940 indicated that the intake of lead appeared to have decreased slightly since that time (Kehoe 1960b,[185] Schroeder and Balassa 1961).[192] When, under experimental conditions, the daily intake of lead from all sources amounted to 0.5 to 0.6 mg over one year or more, a small amount was retained in normal healthy adults but produced no detectable deviation from normal health. Indirect evidence from industrial workers exposed to known amounts of lead for long periods was consistent with these findings (Kehoe 1947).[183]

Young children present a special case in lead intoxication, both in terms of the tolerated intake and the severity of the symptoms (Chisholm 1964).[180] The most prevalent source of lead poisoning of children up to three years of age has been lead-containing paint still found in some older homes (Byers 1959,[179] Kehoe 1960a[184]).

Because of the narrow gap between the quantities of lead to which the general population is exposed through food and air in the course of everyday life, and the quantities that are potentially hazardous over long periods of time, lead in water for human consumption must be limited to low concentrations.

A long-time intake of 0.6 mg lead per day is a level at which development of lead intoxication is unlikely and the normal intake of lead from food is approximately 0.3 mg/day. Assuming a 2 liter daily consumption of water with 0.05 mg/l lead, the additional daily intake would be 0.1 mg/day or 25 per cent of the total intake.

Recommendation

Because of the toxicity of lead to humans and because there is little information on the effectiveness of the defined treatment process in decreasing lead concentrations, it is recommended that 0.05 mg/l lead not be exceeded in public water supply sources.

MANGANESE

Manganese (Mn) is objectionable in public water supplies because of its effect on taste (Riddick et al. 1958,[196] Cohen et al. 1960[194]), staining of plumbing fixtures, spotting of laundered clothes, and accumulation of deposits in distribution systems. Manganese occurs in the reduced state (Mn^{++}), frequently in ground waters and less frequently in surface waters, since exposure to oxygen in surface waters results in oxidation to much less soluble hydrated manganese oxides (American Water Works Association 1971).[193]

Concentrations of manganese less than 0.05 mg/l are generally acceptable in public water supplies, because the characteristic black stains and deposits of hydrated manganese oxides do not manifest themselves. This is the principal reason for limiting the concentration of soluble manganese (Griffin 1960).[195]

Recommendation

On the basis of user preference and because the defined treatment process can remove oxidized manganese but does little to remove soluble manganese (Mn^{++}), it is recommended that 0.05 mg/l soluble manganese not be exceeded in public water sources.

MERCURY

Mercury (Hg) is distributed throughout the environment. As a result of industrial use and agricultural applications, significant local increases in concentrations above natural levels in water, soils, and air have been recorded (Wallace et al. 1971).[209] In addition to the more commonly known sources of man's mercury contributions, the burning of fossil fuels has been reported as a source of mercury pollution (Bertine and Goldberg 1971,[199] Joensuu 1971[202]).

The presence of mercury in fresh and sea water was reported many years ago (Proust 1799,[204] Garrigou 1877,[201] Willm 1879,[210] Bardet 1913[197]). In Germany, early studies (Stock and Cucuel 1934,[206] Stock 1938[205]) found mercury in tap water, springs, rain water, and beer. In all water the concentration of mercury was consistently less than one μg/l, but the beer occasionally contained up to 15 μg/l. A recent survey (U.S. Department of Interior, Geological Survey 1970)[211] demonstrated that 93 per cent of U.S. streams and rivers sampled contained less than 0.5 μg/l of dissolved mercury.

Aside from the exposure experienced in certain occupations, food, particularly fish, is the greatest contributor to the human body burden of mercury (Study Group on Mercury Hazards 1971).[208] The Food and Drug Administration (FDA) has established a guideline of 0.5 mg/kg for the maximum allowable concentration of mercury in fish consumed by humans, but it has not been necessary for the FDA to establish guidelines for other foodstuffs.

Mercury poisoning may be acute or chronic. Generally, mercurous salts are less soluble in the digestive tract than mercuric salts and are consequently less acutely toxic. For man the fatal oral dose of mercuric salts ranges from 20 mg to 30 mg (Stokinger 1963).[207] Chronic poisoning from inorganic mercurials has been most often associated with industrial exposure, whereas that from the organic derivatives has been the result of accidents or environmental contamination.

On the basis of their effects on man, several of the mercury compounds used in agriculture and industry (such as alkoxyalkyls and aryls) can be grouped with inorganic mercury to which the former compounds are usually metabolized. Alkyl compounds are the derivatives of mercury most toxic to man, producing illness from the ingestion of only a few milligrams. Chronic alkyl mercury poisoning is insidious in that it may be manifest after a few weeks or not until after a few years.

It has been estimated (Bergrund and Berlin 1969)[198] that of the total mercury ingested, more than 90 per cent is absorbed via the gastrointestinal tract when taken in the form of methyl mercury; but only 2 per cent is absorbed if it is in the form of mercuric ion (Clarkson 1971).[200] Human excreta reveal a biological half-life of methyl mercury in man of approximately 70 days (Study Group on Mercury Hazards 1971).[208]

Acute mercury toxicity is characterized by severe nausea, vomiting, abdominal pain, bloody diarrhea, kidney damage, and death usually within ten days. Chronic exposure is characterized by inflammation of mouth and gums, swelling of salivary glands, excessive salivation, loosening of teeth, kidney damage, muscle tremors, spasms of extremities, personality changes, depression, irritability, and nervousness (The Merck Index of Chemicals and Drugs 1960).[203]

Safe levels of ingested mercury can be estimated from data presented in "Hazards of Mercury" (Study Group on Mercury Hazards 1971).[208] From epidemiological evidence, the lowest whole blood concentration of methyl mercury associated with toxic symptoms is 0.2 μg/g, which, in turn, corresponds to prolonged, continuous intake by man of approximately 0.3 mg Hg/70 kg/day. When a safety factor of 10 is used, the maximum dietary intake should be 0.03 mg Hg/person/day (30 μg/70 kg/day). It is recognized that this provides a smaller factor of safety for children. If exposure to mercury were from fish alone, the 0.03 mg limit would allow for a maximum daily consumption of 60 grams (420 g/week) of fish containing 0.5 mg Hg/kg. Assuming a daily consumption of 2 liters of water containing 0.002 mg/l (2 μg/l) mercury, the daily intake would be 4 μg. If 420 g of fish per week containing 0.5 mg Hg/kg plus 2 liters of water daily containing 0.002 mg/l mercury were ingested, the factor of safety for a 70 kg man would be 9. If all of the mercury is not in the alkyl form, or if fish consumption is limited, a greater factor of safety will exist.

Recommendation

On the basis of adverse physiological effects and because the defined water treatment process has little or no effect on removing mercury at low levels, it is recommended that total mercury in public water supply sources not exceed 0.002 mg/l.

NITRATE-NITRITE

Serious and occasionally fatal poisonings in infants have occurred following ingestion of well waters shown to contain nitrate (NO_3^-) at concentrations greater than 10 mg/l nitrate-nitrogen (N). This was first associated with a temporary blood disorder in infants called methemoglobinemia in 1945 (Comly 1945).[212] Since then, approximately 2,000 cases of this disease have been reported from private water supplies in North America and Europe, and about 7 to 8 per cent of the infants affected died (Walton 1951,[223] Sattelmacher 1962,[218] Simon et al. 1964[219]).

High nitrate concentrations are frequently found in shallow wells on farms and in rural communities. These are often the result of inadequate protection from barn yard drainage and from septic tanks (U.S. Department of Health, Education, and Welfare, Public Health Service 1961,[221] Stewart et al. 1967[220]). Increasing concentrations of nitrate in streams from farm tile drainage have been shown in regions of intense fertilization and farm crop production (Harmeson et al. 1971).[214]

Many infants have drunk water with nitrate-nitrogen concentrations greater than 10 mg/l without developing the disease. Many public water supplies in the United States have levels of nitrate that routinely exceed the standard, but only one case of methemoglobinemia (Vigil et al. 1965)[222] associated with a public water supply has thus far been reported. Rationale for degrees of susceptibility to methemoglobinemia have yet to be developed.

The development of methemoglobinemia, largely confined to infants less than three months old, is dependent upon the bacterial conversion of the relatively innocuous nitrate ion to nitrite (NO_2^-). Nitrite absorbed into the blood stream converts hemoglobin to methemoglobin. The altered pigment can then no longer transport oxygen, and the clinical effect of methemoglobinemia is that of oxygen deprivation or suffocation. Older children and adults do not seem to be affected, but Russian research reported methemoglobin in five- to eight-year-old school children where the water nitrate concentrations were 182 mg/l as N (Diskalenko 1968).[213]

Nitrite toxicity is well known, but a no-effect level has not been established. When present in drinking water nitrite would have a more rapid and pronounced effect than nitrate. Concentrations in raw water sources are usually less than 1 mg/l as N, and chlorination to a free chlorine residual converts nitrite to nitrate.

Several reviews and reports (Walton 1951,[223] Sattelmacher 1962,[218] Simon et al. 1964,[219] Winton 1970,[224] Winton et al. 1971[225]) generally pointed to 10 mg/l nitrate-nitrogen in drinking water as the maximum tolerance levels for infants. Sattelmacher (1962)[218] showed 3 per cent of 473 cases of infant methemoglobinemia to be associated with levels of less than 9 mg/l as N. Simon and his associates (1964)[219] found 4.4 per cent of 249 cases to be associated with levels less than 11 mg/l as N. Analyses of available data are hampered by the fact that samples for water analysis are sometimes collected weeks or months after the disease occurs, during which time the concentration of nitrate may change considerably. Hereditary defects, the feeding of nitrate-rich vegetables, or the use of common medicines may increase susceptibility to methemoglobinemia. Winton and his associates (1971)[225] concluded that "there is insufficient evidence to permit raising the recommended limit."

Extensive reviews on methemoglobinemia associated with nitrate and nitrite have been provided by Walton (1951),[223] Miale (1967),[216] and Lee (1970).[215] They described the circumstances that contributed to the susceptibility of infants under three months of age to methemoglobinemia from nitrate. These included (a) the stomach pH in infants, which is higher than that of adults and can permit growth of bacteria that can reduce nitrate to nitrite, and (b) infant gastrointestinal illness that may permit reduction of nitrate to nitrite to occur higher in the intestinal tract.

Methemoglobin is normally present at levels of 1 per cent to 2 per cent of the total hemoglobin in the blood. Clinical symptoms are normally detectable only at levels of about 10 per cent. Methemoglobin in the subclinical range has been generally regarded as unimportant. However, 10 children (ages 12 to 14) were observed to have shown conditioned reflexes to both auditory and visual stimuli, as the result of a drinking water source with 20.4 mg/l nitrate-nitrogen. The average methemoglobin in the blood was 5.3 per cent (Petukhov and Ivanov 1970).[217]

Recommendation

On the basis of adverse physiological effects on infants and because the defined treatment process has no effect on the removal of nitrate, it is recommended that the nitrate-nitrogen concentration in public water supply sources not exceed 10 mg/l.

On the basis of its high toxicity and more pronounced effect than nitrate, it is recommended that the nitrite-nitrogen concentration in public water supply sources not exceed 1 mg/l.

NITRILOTRIACETATE (NTA)

Because of its possible large-scale use, nitrilotriacetate (NTA) should be evaluated in light of chronic low-level exposure via drinking water and its potential for adversely affecting the health of the general population. Although nitrilotriacetic acid, a white crystalline powder, is insoluble in water, the tribasic salt is quite soluble.

NTA has strong affinity for iron, calcium, magnesium, and zinc (Bailar 1956).[226] Its relative affinity for toxic metals such as cadmium and mercury is not presently known, nor have its chelating properties in complex ionic solutions been characterized. Copper and lead concentrations in biologically treated waste water after flocculation with aluminum sulfate (125 mg/l) are a function of the NTA present (Nilsson 1971).[227] No information is available on the toxicity of such chelates. No cases of acute human poisoning by NTA have been reported.

In the natural environment, NTA is biodegraded to CO_2, NO_3, and H_2O, with glycine and ammonia as intermediates (Thompson and Duthie 1968).[228] This appears to occur within four to five days. Degradation is accelerated by biological waste treatment. Conversion of NTA to nitrate is on a 1 to 1 molar basis.

Conclusion

No recommendation concerning NTA is made at this time because of the absence of data on affinity for toxic metals, the absence of adequate toxicity data, and the absence of demonstrable effects on man, and because there is doubt about its potential use as a substitute for phosphates in detergents. Toxicity information should be developed and evaluated to establish a reasonable recommendation prior to its use as a substitute.

ODOR

Odor and taste, which are rarely separable, are the primary means by which the user determines the acceptability of water. The absence of odor is an indirect indication that contaminants such as phenolic compounds are also absent, or nearly so. (See *Phenolic Compounds*, in this section, p. 80) Although odor cannot be directly correlated with the safety of the water supply, its presence can cause consumers to seek other supplies that may in fact be less safe.

Many odor-producing substances in raw water supplies are organic compounds produced by microorganisms and by human and industrial wastes (Silvey 1953,[232] Rosen 1966,[231] American Water Works Association, Committee on Tastes and Odors, 1970[230]). The defined treatment process can aid in the removal of certain odorous substances (American Water Works Association 1971),[229] but, it may in other cases increase the odor (Silvey et al. 1950)[233] as by the chlorination of phenolic compounds explained on p. 80.

Recommendation

For aesthetic reasons, public water supply sources should be essentially free from objectionable odor.

OIL AND GREASE

Oil and grease, as defined by Standard Methods (1971),[241] occurring in public water supplies in any quantity cause taste, odor, and appearance problems (Braus et al. 1951,[235] Middleton and Lichtenberg 1960,[240] Middleton 1961a,[239] American Water Works Association 1966[234]), can be hazardous to human health (The Johns Hopkins University, Department of Sanitary Engineering and Water Resources 1956,[237] McKee and Wolf 1963[238]), and are detrimental to the defined treatment process (Middleton and Lichtenberg 1960).[240] Even small quantities of oil and grease can produce objectionable odors and appearance, causing rejection of the water supply before health or treatment problems exist (Holluta 1961,[236] McKee and Wolf 1963).[238]

Recommendation

On the basis of odor and other aesthetic considerations affecting user preference and because oil and grease are unnatural ingredients in water, it is recommended that public water supply sources be essentially free from oil and grease.

ORGANICS-CARBON ADSORBABLE

Organics-carbon adsorbable are composed of carbon-chloroform extract (CCE) (Middleton 1961b)[245] and carbon-alcohol extract (CAE) (Booth et al. 1965,[243] Standard Methods 1971[248]). CCE is a mixture of organic compounds that can be adsorbed on activated carbon and then desorbed with chloroform (Booth et al. 1965).[243] Middleton and Rosen (1956)[246] showed the presence of substituted benzene compounds, kerosene, polycyclic hydrocarbons, phenylether, acrylonitrile, and insecticides in CCE materials. CAE is a mixture of organic compounds that can be adsorbed on activated carbon, then desorbed with ethyl alcohol after the chloroform soluble organics have been desorbed (Booth et al. 1965).[243]

Hueper and Payne (1963)[244] showed that CCE materials had carcinogenic properties when ingested by rats. This study also suggested a life-shortening effect in rats fed CAE materials (Federal Water Pollution Control Administration *Office memorandum* 1963).[249] The CAE material also contained at least one synthetic organic, alkyl benzene sulfonate (Rosen et al. 1956).[247]

It is important to recognize that the carbon usually does not adsorb all organic material present, nor is all the adsorbed material desorbed.

Organics-carbon adsorbable recommendations represent a practical measure of water quality and act as a safeguard against the intrusion of excessive amounts of ill-defined potentially toxic organic material into water. They have served in the past as a measure of protection against the presence of otherwise undetected toxic organic materials in drinking water. However, they provide a rather incomplete index of the health significance of such materials in potable waters.

In 1965 Booth and his associates (1965)[243] developed a Carbon Adsorption Method (CAM) similar to the High-Flow CAM Sampler but with a longer contact time between the sample and the activated carbon. This sampler, called the Low-Flow CAM Sampler, increased organic adsorption and therefore overall yield of the determination.

Since that time a more reliable collection apparatus, called the Mini-Sampler, has been developed (Beulow and Carswell 1972).[242] In addition, the Mini-Sampler also used a type of coal-based activated carbon that enhanced organic collection. Further, the extraction apparatus has been miniaturized to be less expensive and more convenient, and the procedure modified to be more vigorous, thereby increasing desorption and organic recovery (Beulow and Carswell 1972).[242] However, the Mini-Sampler has not been evaluated using raw waters at this time. Therefore, the Low-Flow Sampler (Booth et al. 1965)[243] was used for establishing the recommendation.

Adjustment of the High-Flow Sampler data (1961 Interstate Carrier Surveillance Program) to make them comparable to the recent results from the Low-Flow Sampler show that waters with concentrations exceeding either 0.3 mg CCE/l or 1.5 mg CAE/l may contain undesirable and unwarranted components and represent a generally unacceptable level for unidentified organic substances.

Recommendation

Because large values of CCE and CAE are aesthetically undesirable and represent unacceptable levels of unidentified organic compounds that may have adverse physiological effects, and because the defined treatment process has little or no effect on the removal of these organics, it is recommended that organics-carbon adsorbable as measured by the Low-Flow Sampler (Standard Methods 1971[248]) not exceed 0.3 mg/l CCE and 1.5 mg/l CAE in public water supply sources.

PESTICIDES

Pesticides include a great many organic compounds that are used for specific or general purposes. Among them are chlorinated hydrocarbons, organophosphorus and carbamate compounds, as well as the chlorophenoxy, and other herbicides. Although these compounds have been useful in improving agricultural yields, controlling disease vectors, and reducing the mass growth of aquatic plants in streams and reservoirs, they also create both real and presumed hazards in the environment.

Pesticides differ widely in chemical and toxicological characteristics. Some are accumulated in the fatty tissues of the body while others are metabolized. The biochemistry of the pesticides has not yet been completely investigated. Because of the variability in their toxicity to man and their wide range of biodegradability, the different groups of pesticides are considered separately below.

Determining the presence of pesticides in water requires expensive specialized equipment as well as specially trained personnel. In smaller communities, it is not routine to make actual quantitative determinations and identifications. These are relegated to the larger cities, federal and state agencies, and private laboratories that monitor raw waters at selected locations.

CHLORINATED HYDROCARBON INSECTICIDES

The chlorinated hydrocarbons are one of the most important groups of synthetic organic insecticides because of their number, wide use, great stability in the environment, and toxicity to certain forms of wildlife and other nontarget organisms. If absorbed into the human body, some of the chlorinated hydrocarbons are not metabolized rapidly but are stored in fatty tissues. The consequences of such storage are presently under investigation (Report of the Secretary's Commission on Pesticides and Their Relationship to Environmental Health. U.S. Department of Health, Education, and Welfare 1969).[284] The major chlorinated hydrocarbons have been in use for at least three decades, and yet no definite conclusions have been reached regarding the effect of these pesticides on man (HEW 1969).[284]

Regardless of how they enter organisms, chlorinated hydrocarbons cause symptoms of poisoning that are similar but differ in severity. The severity is related to concentration of the chlorinated hydrocarbon in the nervous system, primarily the brain (Dale et al. 1963).[256] Mild intoxication causes headaches, dizziness, gastrointestinal disturbances, numbness and weakness of the extremities, apprehension, and hyperirritability. In severe cases, there are muscular fasciculations spreading from the head to the extremities, followed eventually by spasms involving entire muscle groups, leading in some cases to convulsions and death.

Very few long term studies have been conducted with human volunteers. The highest level tested for dieldrin was 0.211 mg/man/day for 2 years with no observed illness (Hunter and Robinson 1967,[268] Hunter et al. 1969).[269] Since aldrin is metabolized to dieldrin and has essentially the same toxicity as dieldrin, these data can also be applied to aldrin.

Methoxychlor levels of 140 mg/man/day produced no illness in subjects over a period of 8 weeks (Stein et al. 1965).[280] The maximum level of DDT seen to have no apparent ill effect was 35 mg/man/day for 2 years (Hayes et al. 1971).[265]

The dosage is one of the most important factors in extrapolating to safe human exposure levels. Using tumor-susceptible hybrid strains of mice, significantly increased incidences of tumors were produced with the administration of large doses of DDT (46.4 mg/kg/day) (Innes et al. 1969).[270] In a separate study in mice extending over five generations, a dietary level of 3 ppm of DDT produced a greater incidence of malignancies and leukemia beginning in the second filial and third filial generations, respectively, and continuing in the later generations (Tarjan and Kemeny 1969).[281] These results are preliminary in nature and require confirmation. The findings of both of these studies conflict with earlier studies of the carcinogenic effect of DDT which yielded generally negative results.

A summary of the levels of several chlorinated hydrocarbons that produced minimal toxicity or no effects when fed chronically to dogs and rats is shown in Table II-3 (Lehman 1965,[272] Treon and Cleveland 1955,[282] Cole *unpublished data* 1966[286]). Limits for chlorinated hydrocarbons in drinking water have been calculated primarily on the

TABLE II-3—Recommended Limits for Chlorinated Hydrocarbon Insecticides

Compound	Long-term levels with minimal or no effects					Calculated maximum safe levels from all sources of exposure			Intake from diet		Water	
	Species	ppm in diet	Reference	mg/kg body weight/day^{a}	Reference	Safety Factor (X)	mg/kg/day	mg/man/day^{b}	mg/man/day (8)	% of Safe level	% of Safe level	Recommended limit (mg/l)^{c}
Aldrin	Rat	0.5	(1)	0.083		1/100	0.00083	0.0581	0.0007	5.0	20	0.001
	Dog	1.0	(1)	0.02		1/100	0.0002	0.014^{d}				
	Man			0.003	(2), (3)	1/10	0.0003	0.021				
Chlordane	Rat	2.5	(1)	0.42		1/500	0.00084	0.588^{d}	T	T	5	0.003^{e}
	Dog	N.A.		N.A.								
	Man	N.A.		N.A.								
DDT	Rat	5.0	(1)	0.83		1/100	0.008	0.56^{d}	0.021	3.4	20	0.05
	Dog	400.0	(1)	8.0		1/100	0.08	5.6				
	Man			0.5	(4)	1/10	0.05	3.5				
Dieldrin	Rat	0.5	(1)	0.083		1/100	0.00083	0.0581	0.0049	35.0	20	0.001
	Dog	1.0	(1)	0.02		1/100	0.0002	0.014^{d}				
	Man			0.003	(2), (3)	1/10	0.0003	0.021				
Endrin	Rat	5.0	(5)	0.83		1/500	0.00166	0.1162	0.00035	4.1	20	0.0005
	Dog	3.0	(6)	0.06		1/500	0.00012	0.0084^{d}				
	Man	N.A.		N.A.								
Heptachlor	Rat	0.5	(1)	0.083		1/500	0.000166	0.1162	0.00007	0.6	2	0.0001^{f}
	Dog	4.0	(1)	0.08		1/500	0.00016	0.0112^{d}				
	Man	N.A.		N.A.								
Heptachlor Epoxide	Rat	0.5	(1)	0.083		1/500	0.000166	0.01162	0.0021	150.0	5	0.0001
	Dog	0.5	(1)	0.01		1/500	0.00002	0.0014^{d}				
	Man	N.A.		N.A.								
Lindane	Ra.	50.0	(1)	8.3		1/500	0.0166	1.162	0.0035	8.3	20	0.005
	Dog	15.0	(1)	0.3		1/500	0.0006	0.042^{d}				
	Man	N.A.		N.A.								
Methoxychlor	Rat	100.0	(1)	17.0		1/100	0.17	11.9^{d}	T	T	20	1.0
	Dog	4000.0	(1)	80.0		1/100	0.8	56.0				
	Man			2.0	(7)	1/10	0.2	14.0				
Toxaphene	Rat	10.0	(1)	1.7		1/500	0.0034	0.238^{d}	T	T	2	0.005^{e}
	Dog	400.0	(1)	8.0		1/500	0.016	1.12				
	Man	N.A.		N.A.								

LEGEND: ^{a} Assume weight of rat=0.3 kg and of dog=10 kg; assume average daily food consumption of rat=0.05 kg and of dog=0.2 kg.
^{b} Assume average weight of human adult=70 kg.
^{c} Assume average daily intake of water for man=2 liters
^{d} Chosen as basis on which to derive recommended limit.
^{e} Adjusted for organoleptic effects.
^{f} Adjusted for interconversion to H. epoxide.
N.A. No data available.
T Infrequent occurrence in trace quantities

REFERENCES: (1) Lehman (1965)[272]
(2) Hunter & Robinson (1967)[illegible]
(3) Hunter et al. (1969)[illegible]
(4) Hayes et al. (in press)[illegible]
(5) Treon et al. (1955)[illegible]
(6) Cole (1966)[illegible]
(7) Stein et al. (1965)[illegible]
(8) Duggan and Corneliussen (1972)[illegible]

basis of the extrapolated human intake that would be equivalent to that causing minimal toxic effects in mammals (rats and dogs). For comparison, the dietary levels are converted to mg/kg body weight/day. Aldrin, dieldrin, endrin, heptachlor epoxide, and lindane had lower minimal effect and no-effect levels in dogs than in rats; whereas for DDT, methoxychlor, and toxaphene the converse was observed. Heptachlor was equally toxic to both species. Only data from studies using rats were available for chlordane.

Such data from human and animal investigations have been used to derive exposure standards, as for drinking water, by adjusting for factors that influence toxicity such as inter- and intra-species variability, length of exposure, and extensiveness of the studies. To determine a safe exposure level for man, conventionally, a factor of 0.1 is applied to human data where no effects have been observed; whereas 0.01 is applied to animal data when adequate human data are available for corroboration. A factor of 1/500 is generally used on animal data when no adequate and comparable human data are available.

Thus the human data for aldrin, dieldrin, DDT, and methoxychlor are adjusted by 0.1, and the corresponding animal data for these agents are adjusted by 0.01. The minimal effect levels of chlordane, endrin, heptachlor, heptachlor epoxide, lindane, and toxaphene are adjusted by 1/500; since no adequate human data are available for comparison. These derived values are considered the maximum safe exposure levels from all sources. Because these values are expressed as mg/kg/day, they are readjusted for body weight to determine the total quantity to which persons may be safely exposed.

Analysis of the maximum safe levels (mg/man/day) in Table II-3 reveals that these levels are not exactly the same when one species is compared with another. The choice of level on which to base a level for water requires selection of the lowest value from animal experimentation, provided that the human data are within the same order of magnitude and substantiate that man is no more sensitive to a particular agent than is the rat or the dog.

To then calculate a limit for water it is necessary to consider the exposure from other media. In the case of the chlorinated hydrocarbons, exposure is expected to occur mostly through the diet, although aerial spray of these agents can occasionally result in an inhalation exposure. Dietary intake of pesticide chemicals from 1964 to 1970 have been determined by investigators of the Food and Drug Administration from "market basket" samples of food and water (Duggan and Corneliussen 1972).[258] The average intakes (mg/man/day) are listed in Table II-3.

If the intake from the diet is compared with what are considered acceptable safe levels for these pesticides, it is apparent that only traces of chlordane, methoxychlor, and toxaphene are present in the diet. Less than 10 per cent of the maximum safe level of aldrin, DDT, endrin, heptachlor, or lindane are ingested with the diet. For dieldrin, approximately 35 per cent of the safe level comes from the diet. By contrast, exposure to heptachlor epoxide via the diet accounts for more than the defined safe level. In general, an apportionment to water of 20 per cent of the total acceptable intake is reasonable. However, the limits for chlordane and toxaphene were lowered because of organoleptic effects at concentrations above 0.003 and 0.005 mg/l, respectively (Cohen et al. 1961,[253] Sigworth 1965[278]). The limit for heptachlor epoxide was lowered to five per cent of the safe level because of the relatively high concentrations in the diet; and, accordingly, the limit for heptachlor was lowered because it is metabolized to heptachlor epoxide.

These limits reflect the amounts that can be ingested without harm to the health of the consumer and without adversely affecting the quality of the drinking water. They are meant to serve only in the event that these chemicals are inadvertently present in the water and do not imply that their deliberate addition is acceptable.

Recommendation

Because of adverse physiological effects on humans or on the quality of the water and because there is inadequate information on the effect of the defined treatment on removal of chlorinated hydrocarbons, it is recommended that the limits for water shown in Table II-3 not be exceeded.

ORGANOPHOSPHORUS AND CARBAMATE INSECTICIDES

The number of organophosphorus and carbamate insecticides has steadily increased through special uses in agricultural production and the control of destructive insects. At present, there are perhaps 30 commonly used organophosphates with parathion among those potentially most dangerous to human health. No evidence has developed of any significant contamination of water supplies even in the geographical areas where the use of pesticides in this class has been extensive. However, because of their high mammalian toxicity, it is advisable to establish an upper limit for these pesticides in treated water supplies.

The majority of organophosphorus insecticides in use at present are somewhat similar in chemical structure and in physical and biological properties. Although their specific chemical compositions differ from one another and from carbamates, they all act by the same physiological mechanism. Their presence in public water supplies as contaminants would result in some deleterious biological effect over a period of time.

Ingestion of small quantities of either of these pesticides over a prolonged period results in a dysfunction of the cholinesterase of the nervous system (Durham and Hayes 1962).[259] This appears to be the only important manifestation of acute or chronic toxicity caused by these compounds (HEW 1969).[284]

Although safe levels of these agents have been determined for experimental animals on the basis of biochemical indicators of injury, more knowledge is needed to make specific recommendations for water quality (HEW 1969).[284]

Indications of the levels that would be harmful are available for some organophosphorus compounds as a result of studies conducted with human volunteers. Grob (1950)[263] estimated that 100 mg of parathion would be lethal and that 25 mg would be moderately toxic. On the other hand, Bidstrup (1950)[251] estimated that a dose of 10 to 20 mg of parathion might be lethal. Edson (1957)[260] found that parathion ingested by man at a rate of 3 mg/day had no effect on cholinesterase. Similar values were determined by Williams and his associates (1958).[285] Moeller and Rider (1962)[273] suggested that the detectable toxicity threshold, as measured by cholinesterase depression, was 9 mg/day for parathion equivalency and 24 mg/day for malathion. These investigators also reported that a daily dose of 7 mg of methyl parathion was near the detectable toxicity threshold for this compound; but it was later found (Rider and Moeller 1964)[276] that 10 mg/day of methyl parathion did not produce any significant inhibition of blood cholinesterase. Therefore, 5 mg/day (0.07 mg/kg/day) of parathion equivalency should be a safe intake acceptable to the body.

Frawley and his associates (1963)[262] found that a depression of plasma cholinesterase occurred in human subjects at a dosage of 0.15 mg/kg/day of Delnav, which would amount to a total dose of about 7 to 10 mg/day of parathion depending on the body weight of the subjects.

On the basis that carbamate and organophosphorus insecticides have similar toxic effects and that parathion is one of the most toxic of these classes, the data appeared to show that 0.07 mg/kg/day should be a safe level for the human body. Assuming a daily consumption of 2 liters of water containing cholinergic organophosphates or carbamates in concentrations of 0.1 mg/l, 0.2 mg/day would be ingested. This would provide a factor of safety of 25 for parathion for a man weighing 70 kg.

Recommendation

It is recommended that the carbamate and organophosphorus pesticides in public water sup-

TABLE II-4—Recommended Allowable Levels for Chlorophenoxy Herbicides

Compound	Lowest long-term levels with minimal or no effects			Calculated maximum safe levels from all sources of exposure			Water	
	Species	Dose mg/kg/day[a]	Reference	Safety factor (X)	mg/kg/day	mg/man/day[b]	~% of Safe Level	Rec. limit (mg/l)[c]
2,4-D	Rat	0.5	Lehman (1965)[272]	1/500	0.001	0.07[d]	50	0.02
	Dog	8.0	Lehman (1965)[272]	1/500	0.016	1.12		
2,4,5-TP (Silvex)	Rat	2.6	Mullison (1966)[274]	1/500	0.005	0.35	50	0.03
	Dog	0.9	Mullison (1966)[274]	1/500	0.002	0.14[d]		
2,4,5-T	Rat	4.6	Courtney et al. (1970)[254]	1/1000	0.005	0.35[d]		
			Courtney & Moore (1971)[255]				1	0.002
	Dog	10.0	Drill & Hiratzka (1953)[257]	1/1000	0.01	0.7		

[a] Assume weight of rat = 0.3 kg and of dog = 10 kg; assume average daily food consumption of rat = 0.05 kg and of dog = 0.2 kg.

[b] Assume average weight of human adult = 70 kg.

[c] Assume average daily intake of water for man = 2 liters.

[d] Chosen as basis on which to derive recommended level.

ply sources not exceed 0.1 mg/l, total, because there is inadequate information on the effect of the defined treatment process on their removal.

CHLOROPHENOXY HERBICIDES

During the past 20 years, numerous reservoirs have been constructed as public water supplies for cities and communities in the United States. In certain areas as much as five per cent per year of the total volume of a reservoir may be lost because of the marginal growth of weeds and trees. This is especially common in the Southwest where water levels fluctuate (Silvey 1968).[279]

In recent years the control of aquatic vegetation has been widely practiced for water supply sources in many communities in the U.S. Since herbicides may be used for this purpose, it is possible that some may find their way into finished water.

Two of the most widely used herbicides are 2,4-D (2,4-dichlorophenoxyacetic acid), and 2,4,5-TP (2,4,5-trichlorophenoxy-propionic acid) (see Table II-4). Each of these compounds is available in a variety of salts and esters that may have marked differences in herbicidal properties but are rapidly hydrolyzed to the corresponding acid in the body. There are additional compounds that have been employed from time to time, such as diquat (1,1′-ethylene-2,2′-dipyridylium dibromide) and endothal (disodium 3,6-endoxohexa-hydrophthalate).

Studies of the acute oral toxicity of the chlorophenoxy herbicides indicated that there was approximately a three-fold variation between the species studied and that the acute toxicity was moderate (Hill and Carlisle 1947,[267] Lehman 1951,[271] Drill and Hiratzka 1953,[257] Rowe and Hymas 1954).[277] It appears that acute oral toxicity of the three compounds is of about the same magnitude within each species. In the rat, the oral LD50 for each agent was about 500 mg/kg.

There are some data available on the toxicity of 2,4-D to man indicating that a daily dosage of 500 mg (about 7 mg/kg) produced no apparent ill effects in a volunteer over a 21-day period (Kraus *unpublished* 1946).[288]

Sixty-three million pounds of 2,4-D were produced in 1965. There were no confirmed cases of occupational poisoning and few instances of any illness due to ingestion (Hayes 1963,[264] Nielson et al. 1965[276]) One case of 2,4-D poisoning in man has been reported recently (Berwick 1970).[250]

Lehman (1965)[272] reported that the no-effect level of 2,4-D is 0.5 mg/kg/day in the rat and 8.0 mg/kg/day in the dog. In 2-year feeding studies with the sodium and potassium salts of silvex, the no-effect levels were 2.6 mg/kg/day in rats and 0.9 mg/kg/day, respectively, in dogs (Mullison 1966).[274]

Terata and embryo toxicity effects from 2,4,5-T were evidenced by statistically increased proportions of abnormal fetuses within the litters of mice and rats (Courtney et al. 1970).[254] The rat appeared to be more sensitive to this effect. A dosage of 21.5 mg/kg produced no harmful effects in mice, while a level of 4.6 mg/kg caused minimal but statistically significant effects in the rat. More recent work has indicated that a contaminant (2,3,7,8-tetrachlorodibenzo-p-dioxin) which was present at approximately 30 ppm in the 2,4,5-T formulation originally tested was highly toxic to experimental animals and produced fetal and maternal toxicity at levels as low as 0.0005 mg/kg. However, highly purified 2,4,5-T has also produced teratogenic effects in both hamsters and rats at relatively high dosage rates (FDA and NIEHS *unpublished data*,[287] Collins and Williams 1971[252]). Current production samples of 2,4,5-T that contain less than 1 ppm of dioxin did not produce embryo toxicity or terata in rats at levels as high as 24 mg/kg/day (Emerson et al. 1970).[261]

Recommendation

Because of possible adverse physiological effects and because there are inadequate data on the effects of the defined treatment process on removal of chlorophenoxy herbicides, it is recommended that 2,4-D not exceed 0.02 mg/l, that Silvex not exceed 0.03 mg/l, and that 2,4,5-T not exceed 0.002 mg/l in public water supply sources.

pH

The pH of a raw water supply is significant because it affects water treatment processes and may contribute to corrosion of waterworks structures, distribution lines, and household plumbing fixtures. This corrosion can add such constituents as iron, copper, lead, zinc, and cadmium to the water. Most natural waters have pH values within the range of 5.0 to 9.0. Adjustment of pH within this range is relatively simple, and the variety of anticorrosion procedures currently in use make it unnecessary to recommend a more narrow range.

Recommendation

Because the defined treatment process can cope with natural waters within the pH range of 5.0 to 9.0 but becomes less economical as this range is extended, it is recommended that the pH of public water supply sources be within 5.0 to 9.0.

PHENOLIC COMPONDS

Phenolic compounds are defined (Standard Methods 1971)[301] as hydroxy derivatives of benzene and its condensed nuclei. Sources of phenolic compounds are industrial waste water discharges (Faust and Anderson 1968),[292] domestic sewage (Hunter 1971),[296] fungicides and pesticides (Frear 1969),[294] hydrolysis and chemical oxidation of organophosphorus pesticides (Gomaa and Faust 1971),[295] hydrolysis and photochemical oxidation of carbamate pesticides (Aly and El-Dib 1971),[289] microbial degradation of phenoxyalkyl acid herbicides (Menzie 1969),[298] and naturally occurring substances (Christman and Ghassemi 1966).[291] Some phenolic compounds are sufficiently resistant to microbial degradation to be transported long distances by water.

Phenols affect water quality in many ways. Perhaps the greatest effect is noticed in municipal water systems where trace concentrations of phenolic compounds (usually less than 1.0 mg/l) affect the organoleptic properties of the drinking water. For example, p-cresol has a threshold order concentration of 0.055 mg/l, m-cresol 0.25 mg/l, and o-cresol 0.26 mg/l (Rosen et al. 1962).[300] Phenol has a threshold odor concentration of 4.2 mg/l (Rosen et al. 1962),[300] whereas the values for the chlorinated phenols are: 2-chlorophenol, 2.0 μg/l; and 4-chlorophenol, 250 μg/l (Burttschell et al. 1959).[290] Generally, phenolic compounds are not removed efficiently by the defined treatment process. Furthermore, municipal waters are postchlorinated to insure disinfection. If phenolic compounds are present in waters that are chlorinated for disinfection, chlorophenols may be formed. The kinetics of this reaction are such that chlorophenols may not appear until the water has been distributed from the treatment plant (Lee and Morris 1962).[297]

2,4-dinitrophenol has been shown to inhibit oxidative phosphorylation at concentrations of 184 and 278 mg/l (Pinchot 1967).[299]

The development of criteria for phenolic compounds is hampered by the lack of sensitive standard analytical techniques for the detection of specific phenolic compounds. Some of the more odorous compounds are the para substituted halogenated phenols. These escape detection by the methodology suggested by Standard Methods (1971)[301] unless the analytical conditions are precisely set (Faust et al. 1971).[293]

Recommendation

Because the defined treatment process may severely increase the odor of many phenolic compounds, it is recommended that public water supply sources contain no more than 1 μg/l phenolic compounds.

PHOSPHATE

Recommendations for phosphate concentrations have been considered but no generally acceptable recommendation is possible at this time because of the complexity of the problem. The purpose of such a recommendation would be twofold:

1. to avoid problems associated with algae and other aquatic plants, and
2. to avoid coagulation problems due particularly to complex phosphates.

Phosphate is essential to all forms of life. In efforts to limit the development of objectionable plant growths, phosphate is often considered the most readily controllable nutrient. Evidence indicates (a) that high phosphate concentrations are associated with eutrophication of waters manifest in unpleasant algal or other aquatic plant growths when other growth-promoting factors are favorable; (b) that aquatic plant problems develop in reservoirs or other standing waters at phosphate values lower than those critical in flowing streams; (c) that reservoirs and other standing waters will collect phosphates from influent streams and store a portion of these within the consolidated sediments; and (d) that initial concentrations of phosphate that stimulate noxious plant growths vary with other water quality characteristics, producing such growths in one geographical area but not in another.

Because the ratio of total phosphorus (P) to that form of phosphorus readily available for plant growth is constantly changing and ranges from two to 17 or more times greater, it is desirable to establish limits for total phosphorus rather than to the portion that may be available for immediate plant use. Most relatively uncontaminated lake districts are known to have surface waters that contain 10 to 30 μg/l total phosphorus as P; in some waters that are not obviously polluted, higher values may occur. Data collected by the Federal Water Pollution Control Administration, Division of Pollution Surveillance, indicate that total phosphorus concentrations exceeded 50 μg/l (P) at 48 per cent of the stations sampled across the nation (Gunnerson 1966).[302] Some potable surface water supplies now exceed 200 μg/l (P) without experiencing notable problems due to aquatic growths. Fifty micrograms per liter of total phosphorus (as P) would probably restrict noxious aquatic plant growths in flowing waters and in some standing waters. Some lakes, however, would experience algal nuisances at and below this level.

Critical phosphorus concentrations will vary with other water quality characteristics. Turbidity and other factors in many of the nation's waters negate the algal-producing effects of high phosphorus concentrations. When waters are detained in a lake or reservoir, the resultant phosphorus concentration is reduced to some extent over that in influent streams by precipitation or uptake by organisms and subsequent deposition in fecal pellets or the bodies of dead organisms. At concentrations of complex phosphorus on the order of 100 μg/l, difficulties with coagulation are experienced (U.S. Department of the Interior, Federal Water Pollution Control Administration 1968).[303] (See the discussion of Eutrophication and Nutrients in Section I for a more complete description of phosphorus associations with the enrichment problem.)

Recommendation

No recommendation can be made because of the complexity of relationships between phosphate concentrations in water, biological productivity, and resulting problems such as odor and filtration difficulties.

PHTHALATE ESTERS

Large quantities of phthalate esters are used as plasticizers in plastics. Phthalates in water, fish, and other organisms, represent a potential but largely unknown health problem. They have been implicated in growth retardation, accumulation, and chronic toxicity, but little conclusive information is available (Phthalates are discussed in Section III, Freshwater Aquatic Life and Wildlife.) Because there is insufficient information on their specific effects on man, no scientifically defensible recommendation can be made at this time concerning concentrations of phthalate esters in public water supply sources.

PLANKTON

The quality of public water supplies may be drastically affected by the presence of planktonic organisms. Plankton may be defined as a community of motile or nonmotile microscopic plants and animals that are suspended in water. The species diversity and density of the plankton community are important water quality characteristics that should be monitored in all public water supplies. Several methods for counting plankton have been improvised. Many reports count plankton as number of organisms per aliquot of sample rather than biomass. Since various species of algae are much larger than other species, plankton counts that simply enumerate cells, colonies, or filaments do not indicate accurately the true plankton content of the water (Standard Methods 1971).[305]

Plankters are primarily important in public water supply sources for their contribution to taste and odor problems, pH alteration, or filter clogging. To aid operators in interpreting plankton data, the algae counted should be listed under applicable categories that show the predominance or absence of certain groups of organisms at any given time. The categories used should include green algae, blue-green algae, diatoms, flagellated forms, Protozoa, microcrustaceans and Rotifera, as well as related Protista.

Data from plankton counts can be very useful to water treatment operators (Silvey et al. 1972).[304] Counts of blue-green algae which exceed 50 per cent of the total plankton community usually indicate potential taste and odor problems. So long as the green algae comprise 75 per cent of the total plankton count, it is not likely that serious taste and odor problems will arise. The diatom population of the plankton community is also important. During some diatom blooms, the pH of the water increases enough to require the addition of more alum or iron than would normally be used to achieve the desired pH in the distribution system. Some blooms of planktonic green algae cause the pH of the water to rise from 7.6 to as high as 10. There are apparently no plankters that tend to reduce pH or remove minerals in sufficient quantities to alter conditions.

The role which plankton plays in the productivity of a lake or reservoir is important. The relationship between productivity and respiration may frequently be used as a pollution index. In many instances, plankton studies are more revealing than bacterial studies. A ratio of productivity to respiration amounting to one or more indicates that the algae are producing more oxygen than is being consumed by the bacteria. If the ratio drops below one for significant periods, an undesirable condition exists that may cause problems with anaerobic organisms. For further discussions of productivity and its relation to water quality, see Section I on Recreation and Aesthetics, Section III on Fresh Aquatic Life and Wildlife, and Section IV on Marine Aquatic Life and Wildlife.

POLYCHLORINATED BIPHENYLS (PCB)

Polychlorinated biphenyls (PCB) consist of a mixture of compounds only slightly soluble in water; highly soluble in fats, oils, and nonpolar liquids; and highly resistant to both heat and biological degradation. PCB have a wide variety of industrial uses, primarily as insulating fluid in electrical and heat transfer equipment (Interdepartmental Task Force 1972).[311]

Exposure to PCB is known to cause skin lesions (Schwartz and Peck (1943)[320] and to increase liver enzyme activity that may have a secondary effect on reproductive processes (Risebrough et al. 1968,[317] Street et al. 1969,[321] Wassermann et al. 1970[325]). It is not clear at this time whether the effects are due to PCB or its contaminants, the chlorinated dibenzofurans that are highly toxic (Bauer et al. 1961,[307] Schulz 1968,[319] Verrett 1970[324]). It is also not known whether the chlorinated dibenzofurans are produced by degradation of PCB as well as during its manufacture.

The occurrence of PCB in our waters has been documented repeatedly (*New Scientist* 1966,[315] Holmes et al. 1967,[310] Risebrough et al. 1968,[317] Jensen et al. 1969,[312] Koeman et al. 1969,[313] Schmidt et al. 1971,[318] Veith and Lee 1971[323]). They have been associated with sewage effluents (Holden 1970,[309] Schmidt et al. 1971[318]) and rainwater (Tarrant and Tatton 1968),[322] as well as releases and leakage. Failures of closed systems using PCB have caused some of the more well known releases (Kuratsune et al. 1969,[314] Duke et al. 1970[308]). It has been reported that the defined treatment process does little or nothing to remove PCB (Ahling and Jensen 1970).[306]

An epidemiological study on severe poisoning by rice oil contaminated with polychlorinated biphenyls in 1968 indicated that about 0.5 grams ingested over a period of approximately one month was sufficient to cause the *Yusho* disease. Many of those affected showed no signs of relief after about three years (Kuratsune et al. 1969).[314] Price and Welch (1971)[316] have estimated on the basis of 194 samples that 41 to 45 per cent of the general population of the U.S. may have PCB levels of 1.0 mg/kg or higher (wet weight) in adipose tissue. Therefore, it appears that PCB may accumulate in the body. On this basis it can be calculated that a daily intake of 0.02 mg would require about 70 years to be toxic. Applying a factor of safety of 10 would permit a daily intake of 0.002 mg, and assuming a two liter per day intake, suggests a permissible concentration in water to be 0.001 mg/l.

However, evaluation of the retention and accumulation of PCB from water instead of oil in humans is highly desirable. A study on rats with a single oral dose of 170 mg/kg showed urinary excretion (of PCB) to be limited, while 70 per cent of the dose was found in the feces during an eight week period (Yoshimura et al. 1971).[326] Information on PCB in the diet would also be helpful.

Conclusion

Because too little is known about the levels in waters, the retention and accumulation in humans, and the effects of very low rates of ingestion, no defensible recommendation can be made at this time.

RADIOACTIVITY

The effects of radiation on human beings are viewed as harmful, and any unnecessary exposure to radiation should be avoided. The U.S. Federal Radiation Council* (1961a)[329] provided guidance for federal agencies to limit exposure of individuals to radiation from radioactive materials in the environment. The following statement by the U.S. Federal Radiation Council (1960)[328] is considered especially pertinent in applying the recommendations of this report:

> There can be no single permissible or acceptable level of exposure without regard to the reason for permitting the exposure. It should be general practice to reduce exposure to radiation, and positive effort should be carried out to fulfill the sense of these recommendations. It is basic that exposure to radiation should result from a real determination of its necessity.

The U.S. Federal Radiation Council criteria (1960,[328] 1961a[329]) have been used in establishing the limits for radioactivity recommended here. It should be noted that these guidelines apply to normal peacetime operations. They are predicated upon three ranges of daily intake of radioactivity as seen in Table II-5.

The recommended radionuclide intake derives from the sum of radioactivity from air, food, and water. Daily intakes were prescribed with the provision that dose rates be averaged over a period of one year. The range for specific radionuclides recommended by the U.S. Federal Radiation Council (1961b)[330] are shown in the following tables:

TABLE II-5—Ranges of Transient Rates of Intake (pCi/day) for use in Graded Scale of Action[a]

	Range I	Range II	Range III
Radium-226	0-2	2-20	20-200
Iodine-131[b]	0-10	10-100	100-1000
Strontium-90	0-20	20-200	200-2000
Strontium-89	0-200	200-2000	2000-20,000

[a] See Table II-6.

[b] In the case of iodine-131, the suitable sample would include only small children. For adults, the radiation protection guide for the thyroid would not be exceeded by rates of intake higher by a factor of 10 than those applicable to small children.

TABLE II-6—Graded Scale of Action

Ranges of transient rates of daily intake	Graded scale of action
Range I	Periodic confirmatory surveillance as necessary
Range II	Quantitative surveillance and routine control
Range III	Evaluation and application of additional control measures as necessary

For each range, a measure of control was defined, which represented a graded scale of control procedures.

The U.S. Federal Radiation Council (1961b)[330] further defined the action to be taken by stating that "Routine control of useful applications of radiation and atomic energy should be such that expected average exposures of suitable samples of an exposed population group will not exceed the upper value of Range II." Furthermore, they recommended, with respect to Range III, that "Control actions would be designed to reduce the levels to Range II or lower, and to provide stability at lower levels."

It has not been considered necessary to prescribe criteria for iodine-131 or strontium-89 for surface waters. Iodine-131 has never been a problem in water supplies and does not appear likely to be, and strontium-89 levels should not be significant if strontium-90 levels are kept satisfactorily low. Using the midpoint of Range I, Table II-5, for transient rates of intake recommended by the U.S. Federal Radiation Council, and assuming a 2 liter per day consumption, the radium-226 limit is 0.5 Pc/day and strontium-90 limit is 5 Pc/day. These levels are not currently being exceeded in any surface water supply in the United States, although a number of ground water supplies have more than 0.5 pCi/l of radium-226.

Because tritium (hydrogen-3) may be discharged from nuclear power reactors and fuel reprocessing plants, and because it would not be detected in normal analysis of water samples, it has been considered desirable to include a limit on this low energy radionuclide. The Federal Radiation Council has not provided guidance on tritium intake. A tentative limit of 3,000 pCi/l of tritium has been proposed for the revised edition of Drinking Water Standards. This relatively conservative limit has been suggested because of uncertainty in the potential genetic effects of tritium incorporated into body tissues as tritiated water. It is a generally attainable level based on data from the Environmental Protection Agency Tritium Surveillance System. These data indicate that of 70 United States cities surveyed in 1970, none had an annual average tritium activity in tap water exceeding 3,000 pCi/l, the highest annual average value being 1,900 pCi/l. Levels in surface water collected downstream from nuclear facilities showed only two of 34 locations having tritium activity exceeding 3,000 pCi/l. Precipitation samples taken during 1970 at locations within the United States indicated less than 700 pCi/l.

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* The functions of the U.S. Federal Radiation Council have been transferred to EPA, Office of Radiation Programs.

Although a large number of other radionuclides may be present in water, it has not been considered necessary to include specific limits for other than the three mentioned above. If other nuclides are likely to be present, it is recommended that permissible limits be held to 1/150 of the limit for continuous occupational exposure set by the International Commission on Radiological Protection (1960).[327]

Gross radioactivity limits provide screening techniques and guides to an increased level of radiochemical analysis. If the gross alpha and gross beta concentrations in a sample are less than certain minimum concentrations, no additional radiochemical or radiophysical analyses are required.

Gross Alpha Radioactivity Gross alpha limits or investigation levels are keyed to the concentration limit for radium-226 (the alpha emitter with the most restrictive intake limit). A typical scheme is the following:

TABLE II-7—Typical Scheme of Gross Alpha Concentration

Gross Alpha concentration (pCi/l)	Required action
(a) Not exceeding 0.5 pCi/l	None
(b) Greater than 0.5 but not exceeding 5 pCi/l	Radiochemical analysis for radium-226
(c) Greater than 5 pCi/l	Comprehensive radiochemical anaylsis

Gross Beta Radioactivity Two beta emitting radionuclides with the most restrictive maximum permissible concentrations are lead-210 and radium-228. However, since it is extremely unlikely that either radionuclide will ever be present in a significant concentration in a raw water source, the investigation levels for gross beta radioactivity are keyed to strontium-90 and isotopes of radioiodine.

TABLE II-8—Gross Beta Radioactivity to Strontium-90 and Isotopes of Radioiodine

Gross Beta concentration excluding Potassium-40	Required action
(a) Not greater than 5 pCi/l	None (with knowledge that lead-210 and radium-228 are essentially absent)
(b) Greater than 5, but less than 50 pCi/l	Analyses for strontium-90, iodine-129, and iodine-131
(c) Greater than 50 pCi/l	Comprehensive radiochemical analysis

The radionuclide concentration limits proposed in the above tables should not be considered as absolute maxima that, if exceeded, constitute grounds for rejection of a drinking water supply source. Instead, the concentration limits should be considered guidelines that should not be exceeded unless there is good reason. The constraints that should be imposed are based on: (1) a determination by the appropriate regulatory agencies that the higher level of radioactivity is as low as can be practicably achieved, and (2) quantitative surveillance of all intake pathways to demonstrate that total dose to a suitable sample of the exposed population is within Radiation Protection Guidelines levels. To permit variances in radionuclide concentrations in water depending on concentrations in other environmental media and dietary habits is consistent with the guidance and recommendations of the U.S. Federal Radiation Council, the National Council on Radiation Protection and Measurement, and the International Commission on Radiological Protection.

Recommendation

Because the defined treatment process has uncertain effects on the removal of soluble radionuclides and because of the effects of radiation on humans, it is recommended that the limits related to the guidelines presented above be accepted in the context of the discussion for application to sources of public water supply.

SELENIUM

The toxicity of selenium resembles that of arsenic and can, if exposure is sufficient, cause death. Acute selenium toxicity is characterized by nervousness, vomiting, cough, dyspnea, convulsions, abdominal pain, diarrhea, hypotension, and respiratory failure. Chronic exposure leads to marked pallor, red staining of fingers, teeth and hair, debility, depression, epistaxis, gastrointestinal disturbances, dermatitis, and irritation of the nose and throat. Both acute and chronic exposure can cause odor on the breath similar to garlic (The Merck Index of Chemicals and Drugs 1968).[336] The only documented case of selenium toxicity from a water source, uncomplicated with selenium in the diet, concerned a three-month exposure to well water containing 9 mg/l (Beath 1962).[331]

Although previous evidence suggested that selenium was carcinogenic (Fitzhugh et al. 1944),[332] these observations have not been borne out by subsequent data (Volganev and Tschenkes 1967).[346] In recent years, selenium has become recognized as a dietary essential in a number of species (Schwarz 1960,[341] Nesheim and Scott 1961,[338] Oldfield et al. 1963[339]).

Elemental selenium is highly insoluble and requires oxidation to selenite or selenate before appreciable quantities appear in water (Lakin and Davidson 1967).[335] There is evidence that this reaction is catalyzed by certain soil bacteria (Olson 1967).[340]

No systematic investigation of the forms of selenium in excessive concentrations in drinking water sources has been carried out. However, from what is known of the solubilities of the various compounds of selenium, the principal inorganic compounds of selenium would be selenite and selenate. The ratio of their individual occurrences would depend primarily on pH. Organic forms of selenium occurred in seleniferous soils and had sufficient mobility in an aqueous environment to be preferentially absorbed over selenate in certain plants (Hamilton and Beath 1964).[334] However, the extent to which these compounds might occur in source waters is essentially unknown. Toxicologic examination of plant sources of selenium revealed that selenium present in seleniferous grains was more toxic than inorganic selenium added to the diet (Franke and Potter 1935).[333]

Intake of selenium from foods in seleniferous areas (Smith 1941),[342] may range from 600 to 6,340 μg/day, which approach estimated levels related to symptoms of selenium toxicity in man based on urine samples (Smith et al. 1936,[343] Smith and Westfall 1937[344]). If data on selenium in foods (Morris and Levander 1970)[337] are applied to the average consumption of foods (U.S. Department of Agriculture, Agriculture Research Service, Consumer and Food Economics Research Division 1967),[345] the normal dietary intake of selenium is about 200 μg/day.

If it is assumed that two liters of water are ingested per day, a 0.01 mg/l concentration of total selenium would increase the normal total dietary intake by 10 per cent (20 μg/day). Considering the range of selenium in food associated with symptoms of toxicity in man, this would provide a safety factor of from 2.7 to 29. A serious weakness in these calculations is that their validity depends on an assumption of equivalent toxicity of selenium in food and water, in spite of the fact that a considerable portion of selenium associated with plants is in an organic form. Adequate toxicological data that specifically examine the organic and the inorganic selenium compounds are not available.

Recommendation

Because the defined treatment process has little or no effect on removing selenium, and because there is a lack of data on its toxic effects on humans when ingested in water, it is recommended that public water supply sources contain no more than 0.01 mg/l selenium.

SILVER

Silver is a rather rare element with a low solubility of 0.1 to 10 mg/l depending upon pH and chloride concentration (Hem 1970).[348] Data from 1,577 samples collected from 130 sampling points in the United States showed detectable (0.1 μg/l) concentrations in 104 samples ranging from 1.0 to 38 μg/l with a median of 2.6 μg/l (Kopp 1969).[352]

The principal effect of silver in the body is cosmetic. It causes a permanent grey discoloration of skin, eyes, and mucous membranes. The amounts of colloidal silver required to produce this condition (argyria, argyrosis), which would serve as a basis for determining the water standard, are not known; but the amount of silver from injected agarsphenamine that produces argyria is any amount greater than one gram of silver in the adult (Hill and Pillsbury 1939,[349] 1957[350]). It is also reported that silver, once absorbed, is held indefinitely in the tissues (Aub and Fairhall 1942).[347]

A study that provided analyses of samples of human tissues from 30 normal adult males showed three to contain silver in minute amounts. Comparison of the mean daily concentrations of silver in successive daily samples of urine, feces, and food (0.088 mg/day) showed essentially no absorption of the intake from food (Kehoe et al. 1940b).[351] Studies of the metabolism of silver in the rat showed only about 2 per cent of the element entered the blood from the gastrointestinal tract and that the biological half life was about 3 days (Scott 1949).[353] However, this work was done with carrier free silver and may not be representative of the behavior of larger amounts of element. It does suggest, however, that ingested silver is not likely to be completely stored in the body.

Conclusion

Because silver in waters is rarely detected at levels above 1 μg/l, a limit is not recommended for public water supply sources.

SODIUM

Sodium salts are ubiquitous in the water environment. These minerals are highly soluble, and their concentrations in natural waters show considerable variation, regionally and locally. In addition to natural sources of sodium salts, other sources are sewage, industrial effluents, and deicing salts. Sodium concentrations in ground waters may also vary with well depth, and often reach higher levels of concentration than in surface waters. Removal of sodium is costly and is not common in public water supply treatment.

Of the 100 largest public water supplies in the U.S., most of which are surface supplies, the median sodium content was 12 mg/l with a range of 1.1 mg/l to 177 mg/l (Durfor and Becker 1964).[355] For a healthy individual, the intake of sodium is discretionary and influenced by food selection and seasoning. The intake of sodium may average 6 g/day without adverse effects on health (Dahl 1960).[354]

Various restricted sodium intakes are recommended by physicians for a significant portion of the population, including persons suffering from hypertension, edema associated with congestive cardiac failure, and women with toxemias of pregnancy (National Research Council, Food and Nutrition Board 1954).[356] The sodium intake from sources other than water recommended for very restricted diets is 500 mg/day. Diets for these individuals permit 20 mg/l sodium in drinking water and water used for cooking. If the public water supply has a sodium content exceeding this limit, persons on a very restricted sodium diet must use distilled or deionized water.

For a larger portion of the population who use a moderately restricted diet, 1,000 mg/day is the recommended sodium intake limit (National Research Council, Food and Nutrition Board 1954).[356] Under this limit, water containing a higher concentration of sodium could be used if the sodium intake from the sources other than water were not increased above that of the very restricted diet. Then, the daily intake of sodium from water (20 mg/l for very restricted diets) could be increased by the additional 500 mg (250 mg/l) intake permitted in the moderately restricted diet, thus allowing a significant portion of the population to use public water supplies with higher sodium concentrations. On this basis water containing more than 270 mg/l sodium should not be used for drinking water by those using the moderately restricted sodium diet, and water containing more than 20 mg/l sodium should not be used by those using the very restricted sodium diet.

The response of people who should restrict their sodium intake for health reasons is a continuum varying with intake. The allocation of the difference in dietary intake allowed by the very restricted and the moderately restricted diets to drinking water would be an arbitrary decision. Furthermore, waters containing high concentrations of sodium (greater than 270 mg/l) are likely to be too highly mineralized to be considered desirable from aesthetic standpoints aside from health considerations.

Treatment of an entire public water supply to remove sodium is quite costly. Home treatment for drinking water alone for those needing low sodium water can be done at relatively modest cost, or low sodium content bottled water can be used.

Recommendation

In view of the above discussion no limit is recommended for sodium.

SULFATE

The public water supplies of the 100 largest cities in the United States were found to contain a median sulfate concentration of 26 mg/l, and a maximum of 572 mg/l (Durfor and Becker 1964).[357] Greater concentrations were present in many ground water supplies for smaller communities in the Midwest (Larson 1963).[358] Sulfate ions in drinking water can have a cathartic effect on occasional users, but acclimatization is rapid. If two liters of water are ingested per day, the equivalent sulfate concentrations for laxative doses of Glauber salt and Epsom salt are 300 mg/l and 390 mg/l, respectively (Peterson 1951,[361] Moore 1952[360]).

Data collected by the North Dakota State Department of Health on laxative effects of mineral quality in water indicated that more than 750 mg/l sulfate had a laxative effect, and less than 600 mg/l did not (Peterson 1951).[361] If the water was high in magnesium, the effect took place at lower sulfate concentrations than if other cations were dominant. A subsequent interpretation showed that laxative effects were experienced by sensitive persons not accustomed to the water when magnesium was about 200 mg/l, and by the average person when magnesium was 500–1000 mg/l (Moore 1952).[360]

The median of sulfate concentrations detected by taste by a panel of 10 to 20 persons was 237, 370, and 419 mg/l for sodium, calcium, and magnesium salts, respectively (Whipple 1907).[362] Coffee brewed with 400 mg/l sulfate added as magnesium sulfate was affected in taste (Lockhart et al. 1955).[359]

Recommendation

On the basis of taste and laxative effects and because the defined treatment process does not remove sulfates, it is recommended that sulfate in public water supply sources not exceed 250 mg/l where sources with lower sulfate concentrations are or can be made available.

TEMPERATURE

Temperature affects the palatibility of water by intensifying taste and odor through increased volatility of the source compound (Burnson 1938).[366] Any increase in temperature may stimulate growth of taste and odor producing organisms (Kofoid 1923,[372] Thompson 1944,[378] Silvey et al. 1950[377]) but tends to decrease the survival time of infectious organisms (Peretz and Medvinskaya 1946,[375] Rudolfs et al. 1950[376]). The standard treatment process is also affected by temperature or temperature changes in the steps of coagulation (Velz 1934,[379] Maulding and Harris 1968,[373] American Water Works Association 1971[363]), sedimentation (Camp et al. 1940,[368] Hannah et al. 1967[370]), filtration (Hannah et al. 1967[370]), and chlorination (Ames and Smith 1944,[364] Butterfield and Wattie 1946[367]).

Temperature changes usually are caused by using water as a coolant, as a carrier of wastes, or for irrigation (Brashears, Jr. 1946,[365] Moore 1958,[374] Eldridge 1960,[369] Hoak 1961[371]). Surface water temperatures vary with the seasons, geographical location, and climatic conditions. The same factors along with geological conditions affect ground water temperatures.

Recommendation

No temperature change that detracts from the potability of public water supplies and no temperature change that adversely affects the standard treatment process are suggested guidelines for temperature in public water supply sources.

TOTAL DISSOLVED SOLIDS

(Filterable Residue)

High total dissolved solids (TDS) are objectionable because of possible physiological effects, mineral taste, and economic consequences. Limited research (Bruvold 1967[380]) indicated that consumer acceptance of mineralized waters decreased in direct proportion to increased mineralization. This study covered a range of TDS values of 100 to 1,200 mg/l; one at 2,300 mg/l TDS. For high levels of mineralization, there may also be a laxative effect, particularly upon transients. High concentrations of mineral salts, particularly sulfate and chloride, are also associated with costly corrosion damage in water systems (Patterson and Banker 1968[381]).

Because of the wide range of mineralization of natural water, it is not possible to establish a single limiting value. The measurement of specific conductance provides an indication of the amount of TDS present. The relationship of specific conductance to TDS will vary depending upon the distribution of the major constituent elements present. For any given water a relatively uniform relationship will exist. Where sufficient data exist to establish a correlation between the two measurements, specific conductance may be used as a substitute for the TDS measurement. In very general terms, a specific conductance of 1,500 micro-mhos is approximately equivalent to 1,000 mg/l TDS (Standard Methods 1971).[383]

Because drinking water containing a high concentration of TDS is likely to contain an excessive concentration of some specific substance that would be aesthetically objectionable to the consumer, the 1962 Drinking Water Standards (PHS 1962)[382] included a limit for TDS of 500 mg/l, if other less mineralized sources were available. Although waters of higher concentrations are not generally desirable, it is recognized that a considerable number of supplies with dissolved solids in excess of the 500 mg/l limit are used without any obvious ill effects. Therefore, instead of recommending a general dissolved solids limit, specific recommendations are made in this report for individual substances of importance in drinking water sources, such as chloride and sulfate.

TURBIDITY

The recommendation for acceptable levels of turbidity in water must relate to the capacity of the water treatment plant to remove turbidity adequately and continuously at reasonable cost. Water treatment plants are designed to remove the kind and quantity of turbidity to be expected in each water supply source. Turbidity can reduce the effectiveness of chlorination by physically protecting microorganisms from direct contact with the disinfectant (Sanderson and Kelly 1964,[384] Tracy et al. 1966).[386]

Customary methods (Standard Methods 1971)[385] for measuring and reporting turbidity do not adequately measure those characteristics harmful to public water supply and water treatment processing. A water with 30 turbidity units may coagulate more rapidly than one with 5 or 10 units. Conversely, water with 30 turbidity units sometimes may be more difficult to coagulate than water with 100 units. The type of plankton, clay, or earth particles, their size, and electrical charges, are more important determining factors than the turbidity units. Sometimes clay added to very low turbidity water will improve coagulation.

Turbidity in water should be readily removable by coagulation, sedimentation, and filtration; it should not be present to an extent that will overload the water treatment plant facilities; and it should not cause unreasonable treatment costs. In addition, turbidity should not frequently change or vary in characteristics to the extent that such changes cause upsets in water treatment plant processes.

Conclusion

No recommendation is made, because it is not possible to establish a turbidity recommendation in terms of turbidity units; nor can a turbidity recommendation be expressed in terms of mg/l "undissolved solids" or "nonfilterable solids."

URANYL ION

The 1968 edition of Water Quality Criteria (FWPCA 1968)[387] included a limit for uranyl ion (UO_2^{++}) of 5 mg/l, because a 1965 Public Health Service Drinking Water Standards Review Committee had tentatively decided to include it in the next revision of the Drinking Water Standards. This value was selected because it is below the objectionable taste and appearance levels as well as the chemically toxic concentration.

Further investigation of raw water quality data indicated that uranium does not occur naturally in most waters above a few micrograms per liter (U.S. Geological Survey 1969,[388] EPA *office memorandum* 1971[389]).

Recommendation

The taste, color, and gross alpha recommendations will restrict the uranium concentration to levels below those objectionable on the basis of toxicity. For these reasons, no specific limit is proposed for uranyl ion.

VIRUSES

Many types of viruses are excreted in the wastes of humans and animals (Berg 1971[392]), and some have been implicated in diseases (Berg 1967[391]). There are viruses that alternate between animal hosts (Kalter 1967)[403] and those that can infect genetically distant hosts (Maramorosch 1967).[407] Because almost any virus can be transmitted from host to host through water (Mosley 1967),[409] any amount of virus detectable by appropriate techniques in surface water supplies constitutes a hazard (Berg 1967).[391]

While it is believed that all human enteric viruses have the potential to cause illness in man, not all have been etiologically associated with clinical illness. A number of waterborne local outbreaks attributed to virus affecting approximately 800 people have occurred in the United States, but no obvious large scale spread of a viral disease by the water route is known to have occurred (Mosely 1967).[409] Although virus transmission by water has been suggested for poliomyelitis, gastroenteritis, and diarrhea, the most convincing documentation exists for infectious hepatitis (Mosley 1967).[409] Twelve outbreaks of infectious hepatitis have been attributed to contaminated drinking water in the United States between 1895 and 1971, and most of these have been linked to private systems.

Berg (1971)[392] suggests that waterborne viral disease need not occur at the epidemic level in order to be of significance. Small numbers of virus units could produce infection without causing overt disease, and infected individuals could then serve as sources of larger amounts of virus.

The interpretation of virus data presents other problems in addition to those posed by epidemiological evaluation. There is evidence that one virulent virus unit can be sufficient to infect man if it contacts susceptible cells (Plotkin and Katz 1967),[411] but in an intact host, this is complicated by various defenses (Beard 1967).[390] The interpretation of data is further complicated by aberrations in survival curves for virus thought to be caused by clumping. The statistical treatment of virus data has been discussed by Berg et al. (1967),[393] Chang (1967,[395] 1968[396]), Clark and Niehaus (1967),[399] Sharp (1967),[412] and Berg (1971).[392]

The route of enteric viral contamination of surface waters is from human feces through the effluents of sewage treatment plants as well as contamination from raw sewage. Enteric virus densities in human feces have been estimated by calculation and sampling. Clarke and his associates (1962)[400] suggested that human feces contained approximately 200 virus units per gram per capita and 12×10^6 coliform bacteria per gram per capita, or 15 enteric virus units per 10^6 coliforms. Combining these calculations with observed data, they estimated that sewage contained 500 virus units per 100 ml, and contaminated surface waters contained less than 1 virus unit per 100 ml. These numbers are subject to wide variation and change radically during an epidemic.

The removal capabilities of various sewage treatment processes have been examined individually and in series both in the laboratory and in the field (Chin et al. 1967,[398]

TABLE II–9—Average Time in Days for 99.9 Per Cent Reduction in Original Titer of Indicated Microorganisms at Three Temperatures

Microorganism	Clean water			Moderately polluted water			Sewage		
	28 C	20 C	4 C	28 C	20 C	4 C	28 C	20 C	4 C
Poliovirus 1	17	20	27	11	13	19	17	23	110
ECHO 7	12	16	26	5	7	15	28	41	130
ECHO 12	5	12	33	3	5	19	20	32	60
Coxsackie A9	<8	<8	10	5	8	20	6	No data	12
A. aerogenes	6	8	15	15	18	44	10	21	56
E. coli	6	7	10	5	5	11	12	20	48
S. fecalis	6	8	17	9	18	57	14	26	48

Clarke et al. 1962[400]

Clark and Niehaus 1967,[399] England et al. 1967,[402] Lund and Hedström 1967,[404] Malherbe 1967,[405] Malherbe and Strickland-Cholmley 1967,[406] Berg 1971[392]). These studies indicated that while some sewage treatment processes showed virus removal potential in laboratory tests and field evaluation, there was no indication that consistent adequate virus removal, that is no detectable virus, was accomplished by present sewage treatment practices (Berg 1971).[392] However, the apparent limited survival time for viruses in water can be affected by factors, such as temperature and adsorption that protects viruses; and the proximity of water users may make survival for only a short period of time sufficient to transmit virulent virus (Prier and Riley 1967).[410]

Table II–9 gives virus and bacterial survival data for clean, moderately contaminated, and sewage water.

The removal capabilities of various water treatment processes are presented in Table II–10.

Conventional water treatment processes are variable in their virus removal efficacy and questionable in their performance under field conditions (Berg 1971,[392] Sproul 1972[413]).

Disinfection by chlorination was reviewed recently for its virus inactivation efficacy (Morris 1971).[408] Only undissociated hypochlorous acid (HOCl) was considered effective in virus inactivation. Approximately 25 mg/l chloramine, 100 mg/l hypochlorite or 0.5 to 1.0 mg/l HOCl with 30-minute contact times were required to cause adequate viral inactivation in potable water. The amount of chlorine required to achieve these conditions varied with the pH and the amount of nitrogen present.

TABLE II–10—Removal of Viruses from Water and Wastewater by Biological, Physical, and Chemical Treatment Procedure

Treatment (1)	Menstruum tested (2)	Retention time, in hours (3)	Virus[a] (4)	Virus removed, as a percentage (5)	Reference (6)
Primary setting	Primary effluent	3	Poliovirus 1	0–3	Clarke et al. 1961[401]
Activated sludge	Activated sludge effluent	6.0–8.4	Coxsackievirus A9	96–99	Clarke et al. 1961[401]
		6.0–7.5	Poliovirus 1	88–94[b]	Clarke et al. 1961[401]
Carbon adsorption (0.5 gal per min per sq ft)	Trickling filter effluent		Phage T-2	35	Sproul et al. 1967[414]
$Ca(OH)_2$ coagulation (500 mg per 1)	Activated sludge effluent		Poliovirus 1	98.5–99.9	Berg et al. 1968[394]
$Al_2(SO_4)_3$ coagulation (25 mg per l)	River water		Coxsackievirus A2	95–99[c]	Chang et al. 1958[397]
$FeCl_3$ coagulation (25 mg per l)	River water		Coxsackievirus A2	92–94[c]	Chang et al. 1958[397]

[a] Added to the test experimentally.
[b] When volatile solids were at least 400 mg per l.
[c] When good floc formation occurred.
Berg 1971[392]

Considerable progress on virological method development has been made in the past decade. However, virology techniques have not yet been perfected to a point where they can be used routinely for monitoring water for viruses. There is a need for virus data on relative numbers, better techniques, relative die-off rates, and correlation with existing indicators, as well as methods for direct determination.

Conclusion

In view of the uncertain correlation of virus occurrence with existing indicators, the absence of adequate monitoring techniques, and the general lack of data, scientifically defensible criteria cannot be recommended at this time.

ZINC

Zinc is an essential and beneficial element in human metabolism. The activity of insulin and several body enzymes is dependent on zinc. The daily adult human intake averages 10 to 15 mg; for preschool children it is 0.3 mg/kg. (Vallee 1957).[420]

Zinc is a widely used metal and may be dissolved from galvanized pipe, hot water tanks, or from yellow brass. It may also be present in some corrosion prevention additives and in industrial wastes. The solubility of zinc is variable, depending upon pH and alkalinity.

In 1,577 samples from 130 locations on streams between October 1962 and September 1967, zinc was detected (2 μg/l) in 1,207 samples with a range of 2 to 1,183 μg/l and a mean of 64 μg/l (Kopp 1969).[419]

Individuals drinking water containing 23.8 to 40.8 mg/l of zinc experienced no known harmful effects. Communities have reported using water containing 11 to 27 mg/l of zinc without harmful effects (Bartow and Weigle 1932,[416] Anderson et al. 1934[415]). Another report stated that spring water containing 50 mg/l of zinc was used for a protracted period without harm (Hinman, Jr. 1938[418]).

Statistical analysis of taste threshold tests with zinc in distilled water showed that 5 per cent of the observers were able to distinguish between 4.3 mg/l zinc (added as zinc sulfate) and water containing no zinc salts (Cohen et al. 1960[417]). When added as zinc nitrate and as zinc chloride, the detection levels were 5.2 and 6.3 mg/l zinc, respectively. When zinc sulfate or zinc chloride was added to spring water with 460 mg/l dissolved solids, the detection levels for 5 per cent of the observers were 6.8 and 8.6 mg/l zinc, respectively.

Recommendation

Because of consumer taste preference and because the defined treatment process may not remove appreciable amounts of zinc from the source of the supply, it is recommended that the zinc concentrations in public water supply sources not exceed 5 mg/l.

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INTRODUCTION

The biota of a natural aquatic ecosystem is the result of evolutionary processes in the course of which a delicate balance and complex interactions were established among various kinds of organisms and between those organisms and their environment. Some species can live in a wide range of environmental conditions and are found in many different systems throughout the world. Other species are restricted and their distribution is limited to certain habitats or in some cases to only one. Frequently, it is the latter group of species that have been most useful to man. Minor changes in their environments, especially if such changes are rapid, may upset the ecological balance and endanger the species.

Man has the ability to alter—to impair or improve—his environment and that of other organisms. His use of water to dispose of wastes of a technological society and his other alterations of aquatic environments have degraded his water resources. Water pollutants may alter natural conditions by reducing the dissolved oxygen content, by changing the temperature, or by direct toxic action that can be lethal or, more subtly, can affect the behavior, reproduction, and physiology of the organisms. Although a substance may not directly affect a species, it may endanger its continued existence by eliminating essential sources of food and metabolites. Furthermore, conditions permitting the survival of a given organism at one stage of its life may be intolerable at another stage.

This Section evaluates criteria and proposes recommendations that reflect scientific understanding of the relationships between freshwater aquatic organisms and their environment. Anything added to or removed from natural waters will cause some change in the system. For each use of water there are certain water quality characteristics that should be met to ensure the suitability of the water for that use.

The following general recommendations apply to a wide variety of receiving systems and pollutants:

- More stringent methods of control or treatment, or both, of waste inputs and land drainage should be applied to improve water quality as the demand for use increases.
- In recognition of the limitations of water quality management programs, consideration should be given to providing reserve capacity of receiving waters for future use.
- Bioassays and other appropriate tests, including field studies, should be made to obtain scientific evidence on the effect of wastewater discharges on the environment. Test procedures are recommended in this report.
- A survey of the receiving system to assess the impact of waste discharges on the biological community should be made on a regular basis, particularly prior to new discharges. Such surveys especially should cover the seasons most critical to the biological community. Background laboratory data should include bioassays using important local aquatic organisms and associated receiving waters. In addition to the more comprehensive surveys, some form of biomonitoring in the receiving system should be carried out routinely. A suggested list of ecological considerations is included in the section on Biological Monitoring.
- One of the principal goals is to insure the maintenance of the biological community typical of that particular locale or, if a perturbed community exists, to upgrade the receiving system to a quality which will permit reestablishment of that community.

COMMUNITY STRUCTURE AND PROTECTION OF SIGNIFICANT SPECIES

The natural aquatic environment includes many kinds of plants and animals that vary in their life history and in their chemical and physical requirements. These organisms are interrelated in many ways to form communities. Aquatic environments are protected out of recreational and scientific interest, for aesthetic enjoyment, and to maintain certain organisms of special significance as a source of food. There are two schools of thought as to how this can be accomplished. One is to protect the significant species, the assumption being that by so doing, the entire system is protected. The other approach is to protect the aquatic community, the assumption being that the significant species are not protected unless the entire system is maintained.

Community Structure

Because chemical and physical environments are continually changing—sometimes gradually and sometimes catastrophically—many species are necessary to keep the aquatic ecosystems functioning by filling habitats vacated because of the disappearance of other species. Likewise, when one kind of organism becomes extremely abundant because of the disappearance of one or more species, predator species must be available to feed on the overabundant species and keep it from destroying the functioning of the community. In a balanced ecosystem, large populations of a single species rarely maintain themselves over a long time because predators quickly reduce their number.

Therefore, the diverse characteristics of a habitat are necessary to the maintenance of a functioning ecosystem in the process of evolution. In the fossil record are found many species that were more common at one time than they are today and others that have been replaced entirely. If it were not for diverse gene pools, such evolutionary replacement would not have been possible.

Some aquatic environments present unusual extremes in their chemical and physical characteristics. They support highly specialized species that function as ecosystems in which energy flows and materials cycle. If these species are not present and functioning in this manner, such areas may become aesthetically distasteful, as has occurred for example in the alkaline flats of the West and the acid bogs of the Northeast, Midwest, and East.

Rare habitats support rare organisms that become extinct or endangered species if their habitats are impaired or eliminated. In the aquatic world there are many species of algae, fish, and invertebrates that are maintained only in such rare, fragile habitats. Man must understand them if he is to appreciate the process of evolution and the trend of ecological change that brings about drastic alterations to fauna and flora.

Protection of Significant Aquatic Species

An essential objective of freshwater quality recommendations is the protection of fish and other aquatic organisms for sport or commercial harvesting. This does not imply that all other aquatic species will be subject to potential extinction, or that an unaltered environment is the goal to be attained in all cases. The average person is usually interested in only a small number of aquatic species, principally fish; but it remains necessary to preserve, in certain unique or rare areas, a diversified environment both for scientific study and for maintaining species variety.

It is sometimes difficult to justify protection of isolated organisms not used by man unless it can be documented that they are ultimately essential to the production of desirable biota. In some instances it may be that a critical, sensitive species, irreplaceable in the food web of another more important species, is one known only to the biologist. In such instances, protection of the "less important" sensitive species could justifiably determine the water quality recommendation.

Because no single recommendation can protect all important sport and commercial species unless the most sensitive is protected, a number of species must be considered. The most sensitive species provide a good estimate of the range of sensitivity of all species.

ASSIMILATIVE CAPACITY OF FRESHWATER RECEIVING SYSTEMS

Waste discharges do not just go into water but rather into aquatic ecosystems. The capacity of such a system to receive and assimilate waste is determined by the physical, chemical, and biological interactions within the system. Thus the response is a function of the characteristics of both the ecosystem and the nature and quantity of the waste. Understanding the unique characteristics of each ecosystem will enable wise users to develop means to obtain maximum beneficial use with minimal damage to the system. Each aquatic ecosystem is sufficiently unique to require professional ecological advice to define the problems associated with waste discharge into a particular ecosystem. Such a procedure has not been customary in the past, and this has led to some unfortunate consequences, but the practice is becoming increasingly prevalent.

Aquatic systems receive from natural and man-made sources a variety of organic and inorganic materials. These materials through physical, chemical, and biological interaction are transported, rendered, converted, respired, incorporated, excreted, deposited and thus assimilated by the system. However, not all systems can receive and assimilate the same quantity or kinds of waste materials. The capacity of each system to transform waste without damage to the system is a function of the complexity of environmental factors.

Physical factors such as flow velocity, volume of water, bottom contour, rate of water exchange, currents, depth, light penetration, and temperature, govern in part the ability of a system to receive and assimilate waste materials. This ability is a function of the reaeration capability of the system, the physical rendering of wastes, and other physical, chemical, and biological factors. Most flowing systems have a greater reaeration capacity than standing waters. Furthermore, flowing systems are open systems with continual renewal of water, whereas standing waters are closed systems and act as traps for pollutants.

Temperature plays a vital role in the rate of chemical reactions and the nature of biological activities in freshwater and in governing the receiving and assimilative capacity of a system. Most temperate lakes are thermally stratified part of the year, except when there are small differences between surface and bottom temperatures in the spring and fall. As a consequence little exchange occurs between layers during the period of stratification. In organically enriched lakes and reservoirs, depletion of soluble oxygen typically occurs in the bottom layer because there is little or no photosynthesis and little mixing with the oxygen-rich surface layer. As a result, substances are released from the sediments because certain compounds have a much greater solubility in a reduced state.

The unique chemical characteristics of water govern in part the kinds and quantities of waste a system may receive. Some of the important chemical characteristics are hardness, alkalinity, pH (associated with the buffering capacity), and nutrients such as carbon, nitrogen, and phosphorus. Because of synergistic or antagonistic interaction with receiving water, the effects of a waste on a wide variety of receiving systems are hard to predict.

MIXING ZONES

When a liquid discharge is made to a receiving system, a zone of mixing is created. Although recent public, administrative, and scientific emphasis has focused on mixing zones for the dispersion of heated discharges, liquid wastes of all types are included in the following considerations. (For a further discussion of Mixing Zones see Appendix II-A.)

DEFINITION OF A MIXING ZONE

A mixing zone is a region in which a discharge of quality characteristics different from those of the receiving water is in transit and progressively diluted from the source to the receiving system. In this region water quality characteristics necessary for the protection of aquatic life are based on time-exposure relationships of organisms. The boundary of a mixing zone is where the organism response is no longer time-dependent. At that boundary, receiving system water quality characteristics based on long-term exposure will protect aquatic life.

Recommendation

Although water quality characteristics in mixing zones may differ from those in receiving systems, to protect uses in both regions it is recommended that mixing zones be free of substances attributable to discharges or wastes as follows:

- **materials which form objectionable deposits;**
- **scum, oil and floating debris;**
- **substances producing objectionable color, odor, taste, or turbidity;**
- **conditions which produce objectionable growth of nuisance plants and animals.**

GENERAL PHYSICAL CONSIDERATIONS

The mass emission rates of the most critical constituents and their relationship to the recommended values of the material in the receiving water body are normally the primary factors determining the system-degradation potential of an effluent. Prior to establishment of a mixing zone the factors described in Waste Capacity of Receiving Waters (Section IV, pp. 228–232) and Assimilative Capacity (This Section, p. 111) should be considered and a decision made on whether the system can assimilate the discharge without damage to beneficial uses. Necessary data bases may include:

- Discharge considerations—flow regime, volume, design, location, rate of mixing and dilution, plume behavior and mass-emission rates of constituents including knowledge of their persistence, toxicity, and chemical or physical behavior with time.
- Receiving system considerations—water quality, local meteorology, flow regime (including low-flow records), magnitude of water exchange at point of discharge, stratification phenomena, waste capacity of the receiving system including retention time, turbulence and speed of flow as factors affecting rate of mixing and passage of entrained or migrating organisms, and morphology of the receiving system as related to plume behavior, and biological phenomena.

Mathematical models based in part on the above considerations are available for a variety of ecosystems and discharges. (See Appendix II-A.) All such mathematical models must be applied with care to each particular discharge and the local situation.

Recommendation

To avoid potential biological damage or interference with other uses of the receiving system it is recommended that mixing zone characteristics be defined on a case-by-case basis after determination that the assimilative capacity of the receiving system can safely accommodate the discharge taking into consideration the physical, chemical, and biological characteristics of the discharge and the receiving system, the life history and behavior of organisms in the receiving system, and desired uses of the waters.

GENERAL BIOLOGICAL CONSIDERATIONS

Organisms in the water body may be divided into two groups from the standpoint of protection within mixing zones: (1) nonmobile benthic or sessile organisms; (2) weak and strong swimmers.

1. Nonmobile benthic or sessile organisms in mixing zones may experience long or intermittent exposures exceeding recommended values for receiving systems and therefore their populations may be damaged or eliminated in the local region. Minimum damage to these organisms is attained by minimizing exposure of the bottom area to concentrations exceeding levels resulting in harm to these organisms from long-term exposure. This may be accomplished by discharge location and design.

The mixing zone may represent a living space denied the subject organisms and this space may or may not be of significance to the biological community of the receiving system. When planning mixing zones, a decision should be made in each case whether the nonmobile benthic and sessile organisms are to be protected.

Recommendation

To protect populations of nonmobile benthic and sessile organisms in mixing zones it is recommended that the area of their habitat exposed to water quality poorer than recommended receiving system quality be minimized by discharge location and design or that intermittent time-exposure history relationships be defined for the organisms' well-being.

2. Biological considerations to protect planktonic and swimming organisms are related to the time exposure history to which critical organisms are subjected as they are carried or move through a mixing zone. The integrated time exposure history must not cause deleterious effects, including post-exposure effects. In populations of important species, effects of total time exposure must not be deleterious either during or after exposure.

Weak swimmers and drifting organisms may be entrained into discharge plumes and carried through a mixing zone. In determining the time exposure history and responses of the organisms, the possibility of delayed effects, such as death, disease, and increased vulnerability to predation, should be investigated.

Strong swimmers are capable of moving out of, staying out of, or remaining in a mixing zone. Water quality characteristics which protect drifting organisms should also protect migrating fish moving through mixing zones. However, there are some discharges that attract animals into discharge channels and mixing zones where they are vulnerable to death or shock due to short-term changes in water quality, such as rapid temperature fluctuations. This vulnerability should be recognized and occurrences that expose it should be guarded against (see Chlorine, page 189).

Some free-swimming species may avoid mixing zones and as a consequence the reduced living space may limit the population.

Free-swimming species may be attracted to a discharge. Chronic low-level exposure to toxicants may cause death or affect growth, reproduction or migratory instincts, or result in excessive body-burdens of toxicants hazardous for human consumption.

Recommendation

To protect drifting and both weak and strong swimming organisms in mixing zones it is recommended that scientifically valid data be developed to demonstrate that the organisms can survive without irreversible damage, the integrated time-exposure history to be based on maximum expected residence time so that deleterious effects on populations of important species do not occur.

MEETING THE RECOMMENDATIONS

In mixing zones the exposure of organisms to stress is of greater intensity but usually of shorter duration than in the receiving waters, assuming no attraction by the discharge. The objective of mixing zone water quality recommendations is to provide time exposure histories which produce negligible or no effects on populations of critical species in the receiving system. This objective can be met by: (a) determination of the pattern of exposure in terms of time and concentration in the mixing zone due either to activities of the organisms, discharge schedule, or currents affecting dispersion; and (b) determination that delayed effects do not occur.

Protection would be achieved if the time of exposure met the relationship $T/ET(x) \leq 1$ where T is the time of the organism's exposure in the mixing zone to a specified concentration, and ET(x) is the effective time of exposure to the specified concentration, C, which produces (x) per cent response in a sample of the organisms, including delayed effects after extended observation. The per cent response, (x), is selected on the basis of what is considered negligible effects on the total population and is then symbolized ET(25), ET(5), ET(0.1), etc.

Because concentrations vary within mixing zones, a more suitable quantitative statement than the simple relationship $T/ET(x) \leq 1$ is:

$$\frac{T_1}{ET(x) \text{ at } C_1} + \frac{T_2}{ET(x) \text{ at } C_2} + \frac{T_3}{ET(x) \text{ at } C_3} \cdots + \frac{T_n}{ET(x) \text{ at } C_n} \leq 1$$

where the time of exposure of an organism passing through the mixing zone has been broken into increments, T_1, T_2, T_3, etc. The organism is considered to be exposed to concentration C_1 during the time interval T_1, to concentration C_2 during the time interval T_2, etc. The sum of the individual ratios must then not exceed unity. (See caveat below, Short Time Exposure Safety Factors.)

Techniques for securing the above information, application to a hypothetical field situation, comments, caveats, and limitations are expressed in Appendix II–A, Mixing Zones, Development of Integrated Time Exposure Data, p. 403. Tabular data and formulae for summation of short-term effects of heated discharges on aquatic life are provided in the Heat and Temperature discussion, page 151.

SHORT TIME EXPOSURE SAFETY FACTORS

This concept of summation of short-term effects and extrapolation is an approach which tests the applicability of present bioassay methodology and precision and may not be universally applicable to all types of discharges. Conservatism in application should be practiced. When developing the summation of short-term thermal effects data, a safety factor of two degrees centigrade is incorporated. In development of summation of short-term toxicity effects data, a safety factor exists if a conservative physiological or behavioral response is used with effective time of exposure. However, when mortality is the response plotted, an application factor must be incorporated to provide an adequate margin of safety. This factor can most easily be applied by lowering the sum of the additive effects to some fraction of 1 so that the sum of $T_1/(ET(x) \text{ at } C_1) \cdots + T_n/(ET(x) \text{ at } C_n)$ then equals 0.9, or less. The value must be based on scientific knowledge of the organism's behavior and response to the contaminants involved.

Recommendation

When developing summation of short-term exposure effects it is recommended that safety factors, application factors, or conservative physiological or behavioral responses be incorporated into the bioassay or extrapolation procedures to provide an adequate margin of safety.

OVERLAPPING MIXING ZONES

If mixing zones are contiguous or overlap, the formula expressing the integrated time exposure history for single plumes should be adjusted. Synergistic effects should be investigated, and if not found, the assumption may be made that effects of multiple plumes are additive.

Recommendation

When two plumes are contiguous or overlap and synergistic effects do not occur, protection for aquatic life should be provided if the sum of the fractions of integrated time exposure effects for each plume total ≤ 0.5. Alternatively, protection should be provided if the sum of the fractions for both plumes (or more than two contiguous or overlapping plumes) is ≤ 1. (See caveat above, Short Time Exposure Safety Factors.)

INTERIM GUIDELINE

In the event information on summation effects of the integrated time exposure history cannot be satisfactorily provided, a conservative single figure concentration can be used for all parts of the mixing zone until more detailed determinations of the time-exposure relationships are developed. This single, time-dependent median lethal concentration should be subject to the caveats found throughout this Section and Appendix II–A regarding delayed effects and behavioral modifications. Because of the variables involved, the single value must be applied in the light of local conditions. For one situation a 24-hour LC50 might be adequate to protect aquatic life. In another situation a 96-hour LC50 might provide inadequate protection.

CONFIGURATION AND LOCATION OF MIXING ZONES

The time-dependent three dimensional shape of a discharge plume varies with a multitude of receiving system physical factors and the discharge design. While time exposure water quality characteristics within mixing zones are designed to protect aquatic life, thoughtful placement of the discharge and planned control of plume behavior may increase the level of ecosystem protection, e.g., floating the plume on the surface to protect the deep water of a channel; discharging in midstream or offshore to protect biologically-important littoral areas; piping the effluent across a river to discharge on the far side because fish historically migrate on the near side; or piping the discharge away from a stream mouth which is used by migrating species. Such engineering modifications can sometimes accomplish what is necessary to meet biological requirements.

Onshore discharges generally have more potential for interference with other uses than offshore discharges. For example the plume is more liable to impinge on the bottom in shallow areas of biological productivity and be closer to swimming and recreation areas.

PROPORTIONAL RELATIONSHIP OF MIXING ZONES TO RECEIVING SYSTEMS

Recommendations for mixing zones do not protect against the long-term biological effects of sublethal conditions. Thus water quality requirements necessary to protect all life stages and necessary functions of aquatic organisms such as spawning and larval development, are not provided in mixing zones, and it is essential to insure that adequate portions of every water body are free of mixing zones. The decision as to what portion and areas must be retained at receiving water quality values is both a social and scientific decision. In reaching this decision, data input should include current and projected information on types and locations of intakes and discharges; percentage of shoreline necessary to provide adequate spawning, nursery, and feeding areas; and other desired uses of the water.

Recommendation

It is recommended that the total area or volume of a receiving system assigned to mixing zones be limited to that which will: (1) not interfere with biological communities or populations of important species to a degree which is damaging to the ecosystem; (2) not diminish other beneficial uses disproportionately.

ZONES OF PASSAGE

In river systems, reservoirs, lakes, estuaries, and coastal waters, zones of passage are continuous water routes of such volume, area, and quality as to allow passage of free-swimming and drifting organisms so that no significant effects are produced on their populations.

Transport of a variety of organisms in river water and by tidal movements in estuaries is biologically important in a number of ways; e.g., food is carried to the sessile filter feeders and other nonmobile organisms; spatial distribution of organisms and reinforcement of depauperate populations is enhanced; embryos and larvae of some fish species develop while drifting. Anadromous and catadromous species must be able to reach suitable spawning areas. Their young (and in some cases the adults) must be assured a return route to their growing and living areas. Many species make migrations for spawning and other purposes. Barriers or blocks which prevent or interfere with these types of essential transport and movement can be created by water of inadequate chemical or physical quality.

Water quality in the zone of passage should be such that biological responses to the water quality characteristics of the mixing zone are no longer time-dependent (see Definition of Mixing Zone on page 112). However, where a zone of passage is to be provided, bioassays determining time-exposure responses in the mixing zone should include additional requirements to assess organism behavior. In the mixing zone discussion above it is assumed that entrainment in the plume will be involuntary. However, if there is attraction due to plume composition, exposure in the plume could be very much longer than would be predicted by physical modeling. If avoidance reactions occur, migration may be thwarted. Thus, concentrations in both the mixing zone and the zone of passage should be reduced before discharge to levels below those at which such behavioral modifications affect the populations of the subject organisms.

Modern techniques of waste water injection such as diffusers and high velocity jets may form barriers to free passage due to responses of organisms to currents. Turbulence of flows opposing stream direction may create traps for those organisms which migrate upstream by orientation to opposing currents. These organisms may remain in the mixing zone in response to currents created by the discharge.

Recommendation

Because of varying local physical and chemical conditions and biological phenomena, no single-value recommendation can be made on the percentage of river width necessary to allow passage of critical free-swimming and drifting organisms so that negligible or no effects are produced on their populations. As a guideline no more than ⅔ the width of a water-body should be devoted to mixing zones thus leaving at least ⅓ free as a zone of passage.

BIOLOGICAL MONITORING

Monitoring of aquatic environments has traditionally included obtaining physical and chemical data that are used to evaluate the effects of pollutants on living organisms. Biological monitoring has received less emphasis than chemical or physical monitoring, because biological assessments were once not as readily amenable to numerical expression and tended to be more time consuming and more expensive. This is no longer true. Aquatic organisms can serve as natural monitors of environmental quality and should be included in programs designed to provide continuous records of water quality, because they integrate all of the stresses placed on an aquatic system and reflect the combined effect. Chemical-physical assessments identify individual components, so the two types of assessments are mutually supporting rather than mutually exclusive.

A biological monitoring program is essential in determining the synergistic or antagonistic interactions of components of waste discharges and the resulting effects on living organisms. However, biological monitoring does not replace chemical and physical monitoring; each program provides information supplemental to the others.

PROGRAMS

An ideal biological monitoring program has four components: (1) field surveys, (2) in-plant biological monitoring, (3) bioassays, and (4) simulation techniques. Obviously no biological monitoring program is routine, nor does it necessarily have to include all of the above components. However, each of the components provides valuable and useful information.

FIELD SURVEYS

Field surveys are needed to obtain adequate data on biological, chemical, and physical water quality to determine the nature of the system and the possible adverse effects of waste discharges on beneficial uses of the system. Two methods for continuously monitoring the effects of pollution on a receiving water have been described. Patrick et al. (1954)[64]* described the use of diatoms as natural monitors of various types of pollution. Various species of shellfish, especially oysters suspended in trays, have been described as an effective method of monitoring pollution (Galtsoff et al. 1947).[4] Field surveys should be carried out at suitable intervals depending on local conditions. For example, in determining the impact of a new or relocated municipal or industrial discharge, it is desirable to perform the following functions:

- survey the stream as a part of the site selection procedure;
- continue the field survey prior to construction to determine existing water quality: at this time it is also useful to make bioassays using simulated plant wastes and representative organisms from the receiving systems, and to establish biomonitoring stations;
- monitor the effects of construction;
- carry out bioassays using actual plant wastes and effluents after the plant is in operation, and make field surveys to determine any changes from preconstruction results.

BODY BURDENS OF TOXICANTS

Body burdens of toxicants that can be concentrated by biota should be measured regularly. These data can provide early warning before concentrations in water become readily available and can provide warnings of incipient effects in the biota being monitored.

IN-PLANT BIOLOGICAL MONITORING

Present information systems do not provide data rapidly enough to be of use in environmental management, because the constituents of a waste stream are likely to vary from hour to hour and from day to day. Potentially harmful materials should be detected before they enter the receiving water and before substantial damage has been done to the ecosystem.

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* Citations are listed at the end of the Section. They can be located alphabetically within subtopics or by their superior numbers which run consecutively across subtopics for the entire Section.

Several potentially useful methods for rapid in-plant monitoring are being explored (Sparks et al. 1969,[7] Waller and Cairns 1969[8]), and one rapid in-stream method is now operational (Cairns et al. 1968,[2] Cairns and Dickson 1971[3]). These in-plant methods use changes in heart rate, respiration, and movements of fish within a container to detect sublethal concentrations of toxicants in a waste discharge. Continual information on toxicity of a waste should enable sanitary engineers to identify those periods likely to produce the most toxic wastes and to identify those components of the production process that contribute significantly to toxicity. This could be accomplished with bioassays as they are currently used, but rarely are enough samples taken over a period of time sufficient to give the range of information that would be available with continually operating bioassay techniques.

BIOASSAYS

Of equal importance to the river surveys and the in-plant and in-stream monitoring systems is the availability of toxicity information based on a predictive bioassay. The bioassay provides valuable information pertaining to the effects of potential or contemplated discharges on aquatic life. Acute bioassays are useful as a shortcut or predictive method of estimating safe concentrations by use of suitable application factors for many pollutants, as recommended throughout this Report.

However, determining only the acute lethal toxicity of wastes is no longer adequate. Good health and an ability to function vigorously are as important for aquatic ecosystems as they are for humans. The former end point of bioassays, viz., death, has been supplanted by more subtle end points such as the protection of respiration, growth, reproductive success, and a variety of other functional changes (Cairns 1967).[1] Acute toxicity determinations are being supplemented by long-term tests often involving an entire life cycle. The latter require more time and expense than short-term tests, but they provide better predictive information about biologically safe concentrations of various toxicants. Bioassays of organisms other than fish are becoming increasingly common because of the realization that elimination of the lower organisms can also have serious consequences.

SIMULATION TECHNIQUES

The fourth component now available to provide ecological information is the use of scale models. Models are used to study major ecological or environmental problems by simulating prospective new uses. Engineering scale models are common, but ecological scale models or environmental simulation systems are not yet as commonly used. Experimental streams and reservoirs have been constructed to predict toxicity of waste discharges, determine factors responsible for productivity of aquatic communities, and answer questions about plant site location (Haydu 1968,[5] Warren and Davis 1971[9]).

BIOASSAYS

Bioassays are used to evaluate a given pollutant in terms of existing water quality. Most pollution problems involve discharges of unknown and variable composition where more than one toxicant or stress is present. In evaluating criteria for specific toxicants, consideration must be given to other environmental influences such as dissolved oxygen, temperature, and pH.

Harmful effects of pollutants can be described by one or more of the following terms:

acute—involves a stimulus severe enough to bring about a response speedily, usually within four days for fish.

subacute—involves a stimulus less severe than an acute stimulus, producing a response in a longer time; may become chronic.

chronic—involves a lingering or continuous stimulus; often signifying periods of about one-tenth of the life span or more.

lethal—causes death by direct action.

sublethal—insufficient to cause death.

cumulative—brought about, or increased in strength, by successive additions.

Two broad categories of effect (Alderdice 1967)[10] may be distinguished: acute toxicity which is usually lethal, and chronic toxicity which may be lethal or sublethal.

MEASURES OF TOXICITY

Most of the available toxicity data are reported as the median tolerance limit (TLm or TL50) or median lethal concentration (LC50). Either symbol signifies the concentration that kills 50 per cent of the test organisms within a specified time span, usually in 96 hours. The customary 96-hour (four-day) time period is recommended as adequate for most routine tests of acute toxicity with fish. A threshold of acute toxicity will have been attained within this time in the majority of cases (Sprague 1969).[43] This *lethal threshold concentration* is usually noticeable in the data. Sometimes mortality continues, and tests of a week or longer would be necessary to determine the threshold. The lethal threshold concentration should be reported if it is demonstrated, because it is better for comparative purposes than the arbitrary 96-hour LC50. Absence of any apparent threshold is equally noteworthy.

The median lethal concentration is a convenient reference point for expressing the acute lethal toxicity of a given toxicant to the average or typical test animal. Obviously it is in no way a safe concentration, although occasionally the two have been confused. Safe levels, which permit reproduction, growth, and all other normal life-processes in the fish's natural habitat, usually are much lower than the LC50. In this book, the recommended criteria are intended to be safe levels.

Substantial data on long-term effects and safe levels are available for only a few toxicants. Information is now accumulating on the effect of toxicants on reproduction, an important aspect of all long-term toxicity tests. Other information is being gathered on sublethal effects on growth, performance, avoidance reactions, and social behavior of fish. Also important is the sensitivity of organisms at various life stages. Many organisms are most sensitive in the larval, nymphal, molting, or fry stage; some are most sensitive in the egg and sperm stage.

It would be desirable if a single, universal, rapid, biological test could be used to measure directly sublethal effects of a pollutant. Data on sublethal responses of fish have been used, such as respiratory rates and "coughing," swimming speed, avoidance behavior, and specific physiological and biochemical changes in various organisms; and histological studies have been made. A review of these (Sprague 1971)[45] shows that no single test is meaningful for all kinds of pollutants. Therefore, it is recommended that routine assessment and prediction of safe levels be made by carrying out bioassays for acute lethal toxicity and multiplying the lethal concentration by a suitable application factor. The application factors used and recommended here have been derived principally from chronic or sublethal laboratory experiments or from well documented field studies of polluted situations.

Acceptable concentrations of toxicants to which organisms are exposed continually must be lower than the higher concentrations that may be reached occasionally but briefly without causing damage. Both maximum short-time concentrations and the more restrictive range of safe concentrations for continuous exposure are useful. The recommendations in this Report are those considered safe for continuous exposure, although in some cases there has also been an indication of permissible higher levels for short periods.

In field situations and industrial operations, average 24-hour concentrations can be determined by obtaining composite or continuous samples. After 24 hours, the sample may be mixed and analyzed. The concentration found will represent the average concentration. Samples obtained this way are more reproducible and easier to secure than the instantaneous sample of maximum concentrations. However, average concentrations are of little significance if fish are killed by a sharp peak of concentration, and for that reason maximum concentrations must also be considered.

METHODS FOR BIOASSAYS

Although there are many types of assays, two are in general use:

1. the static bioassay in which the organisms are held in a tank containing the test solution, and
2. the continuous flow or flow-through bioassay in which the test solution is renewed continually.

The difference between the two types is not always great, but one can have clear advantages over the other.

An outline of methods for routine bioassays has been given in "Standard Methods for the Examination of Water and Wastewater" (American Public Health Association, American Water Works Association, Water Pollution Control Federation, 1971,[11] hereafter referred to as Standard Methods 1971[48]). Cope (1961)[21] described bioassay reporting, and Cairns (1969)[20] presented a rating system for evaluating the quality of the tests. Sprague (1969,[43] 1970,[44] 1971[45]) reviewed research to develop more incisive testing methods. Their findings are utilized in this Report.

Procedure for acute bioassay with fish is now relatively standardized and usually incorporates:

- a series of replicate test containers, each with a different but constant concentration of the toxicant;
- a group of similar fish, usually 10, in each container;
- observations of fish mortality during exposures that last between one day and one week, usually four days; and
- final results expressed as LC50.

Other factors that are required for good bioassay practice are briefly summarized in the references mentioned above.

CHECKLIST FOR PROCEDURES

Species

A selected strain of fish or other aquatic organisms of local importance should be used in bioassays conducted for the purpose of pollution monitoring. Preferably it should be a game or pan fish, which are usually among the more sensitive. Ability to duplicate experiments is enhanced by the use of a selected strain of test organisms (Lennon 1967).[31] A selected strain can also help to determine the difference between toxicants more reliably, and to detect discrepancies in results due to apparatus. A National Research Council subcommittee chaired by Dr. S. F. Snieszko is currently preparing a report, *Standards and guidelines for the breeding, care, and management of laboratory animals—Fish*, which will be useful in this area. Susceptibility to toxicants among different species of fish is generally less than might be expected—sometimes no greater than when a single species is tested in different types of water. For example, trout and certain coarse fishes were equally resistant to ammonia when tests continued for several days to give the less sensitive species time to react (Ball 1967a);[13] and even for zinc, the coarse fishes were no more than 3.8 times as resistant as trout (Ball 1967b).[14] Recommendations for the selected test fish will often provide protection to other aquatic animals and plants. There are exceptions to this generalization: for example, copper is quite damaging to algae and mollusks, and insecticides are especially dangerous to aquatic arthropods. Sufficient data exist to predict these situations. When they are expected, bioassays should be run with two kinds of invertebrates and two kinds of algae (Patrick et al. 1968).[41]

In the case of important bodies of water, there is good reason to test several kinds of aquatic organisms in addition to fish. Patrick et al. (1968)[41] made a comparative study of the effects of 20 pollutants on fish, snails, and diatoms and found that no single kind of organism was most sensitive in all situations. The short-term bioassay method for fish may also be used for many of the larger invertebrate animals. A greater volume of test water and rate of flow, or both, may be required in relation to weight of the animals since their metabolic rate is higher on a weight basis.

Larvae of mollusks or crustaceans can be good test animals. The crustacean *Daphnia* is a good test animal and was widely used in comparative studies of toxicants by Anderson (1950).[12] Recently Biesinger and Christensen (*unpublished data*, 1971)[52] have carried out tests on the chronic effects of toxicants on growth, survival, and reproduction of *Daphnia magna*. Because of the rapid life cycle of *Daphnia*, experiments on chronic toxicity can be completed in about the same time as an acute toxicity test with fish.

Patrick et al. (1968)[41] have shown that diatoms, snails and fish exposed for roughly comparable periods of time and in similar environmental conditions very often have similar LC50's, but at other times these may differ greatly.

However, for some toxicants diatoms were most sensitive; for others, fish; and for others, snails. When one is comparing data of this type, one questions whether a LC50 for a diatom population in which a number of divisions have occurred during the test period is comparable to that obtained for fish and snails in which no reproduction has occurred during the test period. In the sense that there are 50 per cent fewer cells in the LC50 concentration than there are in the diatom control culture, the test is somewhat equivalent to a test of acute toxicity that results in 50 per cent fewer surviving fish in the LC50 than in the control container. Also loss of ability to grow and divide might be just as fatal to a microbial population as death of a substantial number of its members would be to a fish population.

When the absolute time for the test is considered, there are also reasons for believing that exposure of diatoms to a toxicant through several generations might not constitute a chronic test, because it is quite possible that for toxicants to accumulate in a cell may require a period of exposure much more lengthy than that encompassed in the average test which only spans a few generations. This would be particularly true when the organisms were dividing rapidly and the additional protoplasm diluted the material being accumulated.

Dilution Water

Toxicants should be tested in the water that will receive the pollutant in question. In this way all modifying factors and combined toxicities will be present. It is not advisable to use tap water for dilution, because it may contain chlorine and other harmful materials such as copper, zinc, or lead from plumbing systems. Routine dechlorination does not insure complete removal of chlorine.

Variations in physical and chemical characteristics of water affect toxicity of pollutants. Effects of five environmental entities on the lethal threshold of ammonia were illustrated a decade ago (Lloyd 1961b).[34] Hardness of water is particularly important in toxicity of metals. Hydrogen ion concentration is an important modifying factor for ammonia and cyanide. Higher temperatures sometimes increase toxicity of a pollutant, but recent work shows that phenol, hydrogen cyanide, ammonia, and zinc may be more toxic at low temperatures (United Kingdom Ministry of Technology 1969).[50] Dissolved oxygen levels that are below saturation will increase toxicity, and this is predictable (Lloyd 1961a;[33] Brown 1968).[16]

The supply of dilution water must be adequate to maintain constant test conditions. In both static and continuous flow tests, a sufficiently large volume of test water must be used, and it must be replaced or replenished frequently. This is to provide oxygen for the organism and dilution of metabolic wastes, to limit changes in temperature and pH, and to compensate for degradation, volatilization, intake, and sorption of the toxicant. In static tests, there should be two or three liters of water per gram of fish, changed daily, or increased proportionally in volume for the number of days of the test. In continuous flow tests, the flow must provide at least two or three liters of water per gram of fish per day, and it must equal test-volume in five hours or less, giving 90 per cent replacement in half a day or less.

Acclimation

Acclimatizing the test organism to the specific water before the bioassay begins may have marked effect upon the outcome. Abrupt changes in quality of the water should be avoided. Time for acclimation of the organisms to the conditions of the diluent water should be as generous as possible, dependent on life span. At least two weeks is recommended for fish.

Test Methods

Test methods must be adequately described when the results are given. Several bioassay procedures are listed in Table III–1. Adequate and appropriate control tests must always be run (Sprague 1969).[43] Survival of the control organisms is a minimum indication of the quality of the test organisms. In addition, levels of survival and health in holding tanks should be indicated and the conclusions recorded.

TABLE III–1—Recommended Literature Sources for Bioassay and Biomonitoring Procedures with Various Aquatic Organisms

Kind of organism	Type of response	Appropriate situations for use	Reference
Fish and Macroinvertebrates	96-hour lethal concentration	To measure lethal toxicity of a waste of known or unknown composition. To serve as a foundation for extrapolating to presumably safe concentrations. To monitor industrial effluents.	Standard Methods 1971[48]
Fish and macroinvertebrates	Lethal threshold concentration	For research applications to document lethal thresholds.	Sprague 1969,[43] 1970[44]
Fish and invertebrates	Incipient lethal temperatures & ultimate incipient lethal temperatures	For research to determine lethal temperature ranges of a given species.	Fry 1947,[29] Brett 1952[13]
Fish	Respiratory movements as acute sublethal response	Quick (1-day) indication of possible sublethal effects. For research and monitoring.	Schaumburg et al. 1967[45]
Fish (i.e., fathead minnows, brook trout, bluegill)	Reproduction, growth, and survival	Chronic tests for research on safe concentrations.	Mount 1968,[37] Mount & Stephan 1967,[38] Brungs 1969,[18] McKim & Benoit 1971,[36] Eaton 1970[24]
Daphnia	Survival, growth, and reproduction	Rapid completion of chronic tests for testing special susceptibility of crustaceans	Anderson 1950,[11] Biesinger & Christensen (Unpublished data)[53]
Diatoms	Survival, growth, and reproduction	A sensitive, rapid, chronic test for research, prediction, or monitoring[a]	Patrick 1968[40]
Marine crustaceans, larvae mollusks	Survival, growth, and development through immature stages	A sensitive, rapid, chronic test for research, prediction, or monitoring[a]	Woelke 1967[51]

[a] requires an operator with some specialized biological training.

Dissolved Oxygen

The problem of maintaining dissolved oxygen concentrations suitable for aquatic life in the test water can be difficult. The suggestions on test volume and replacement times (see Dilution Water above) should provide for adequate oxygen in most cases. However, with some pollutants, insufficient oxygen may be present in the test water because a biochemical and a chemical oxygen demand (BOD and COD) may consume much of the available dissolved oxygen. Aeration or oxygenation may degrade or remove the test material. Devices for maintaining satisfactory dissolved oxygen in static tests have been proposed and used with some degree of effectiveness, and are described in Doudoroff et al. (1951).[22]

Concentrations

Periodic measurements of concentration of the toxicant should be made at least at the beginning and end of the bioassay. If this is not possible, introduced concentrations may be stated alone, but it should be realized that actual concentrations in the water may become reduced.

In the flow-through type of bioassay, a large quantity of test water can be made up and used gradually. More often a device is used to add toxicant to a flow of water, and the mixture is discharged into the test container, using apparatus such as "dipping bird" dosers described by Brungs and Mount (1967).[19] Other devices have been developed by Stark (1967),[47] and Mount and Warner (1965),[39] using the doser technique.

Evaluation of Results

Mortality rates at the longest exposure time should be plotted on a vertical probit scale against concentrations of toxicants on a horizontal logarithmic scale. The concentration which causes 50 per cent mortality can be read and used as LC50. Errors in LC50 can be estimated using the simple nomograph procedures described by Litchfield and Wilcoxon (1949).[32] A more refined estimate of error may be made using the methods of Finney (1952),[25] which can be programmed for a computer.

The value of the results would be improved if the LC50's were estimated (by the above procedures) at frequent exposure times such as 1, 2, 4, 8±1, 14±2, 24, 48, 72, and 96 hours. A toxicity curve of time versus LC50 could then be constructed on logarithmic axes. The lethal threshold concentration could then be estimated in many cases (Sprague 1969)[43] to provide a more valid single number for description of acute toxicity than the arbitrary 96-hour LC50.

For some purposes, such as basic research or situations where short exposures are of particular concern, it would be desirable to follow and plot separately the mortality of the group of fish in each tank. In this way, the median lethal time can be estimated for a given concentration. Methods for doing this are given in Appendix II–A.

APPLICATION FACTORS

Short-term or acute toxicity tests do not indicate concentrations of a potential toxicant that are harmless under conditions of long-term exposure. Nevertheless, for each toxicant there is obviously a numerical value for the ratio of the safe concentration to the acutely lethal concentration. Such values are called application factors. In some cases this safe-to-lethal ratio is known with reasonable accuracy from experimental work, as in the examples given in Table III–2. However, for most toxicants, the safe level has not been determined, and must be predicted by some approximate method. In these cases, the assumption has been made in this Report, that the numerical value of the safe-to-lethal ratio, the application factor, is constant for related groups of chemicals. Values for the ratio will be recommended. The safe level of a particular toxicant can then be estimated approximately by carrying out an acute bioassay to determine the lethal concentration, then multiplying this by the suggested application factor. An application factor does not make allowance for unknown factors. It is merely a fractional or decimal factor applied to a lethal concentration to estimate the safe concentration.

Ideally, an application factor should be determined for each waste material in question. To do this, it is necessary first to determine the lethal concentration of the waste according to the bioassay procedures outlined above. To obtain the application factor, the safe concentration of the same waste must be determined for the same species by thorough research on physiological, biochemical, and behavioral effects, and by studying growth, reproduction, and production in the laboratory and field. The safe-to-lethal ratio obtained could then be used as an application factor in a given situation, by working from the measured LC50 of a particular kind of waste to predict the safe concentration.

TABLE III–2—Ratios between the safe concentration and the lethal concentration which have been determined experimentally for potential aquatic pollutants. Sources of data are given in the sections on the individual pollutants.

Material	Species of animal	Safe-to-lethal ratio
LAS	Fathead minnow (Pimephales promelas)	Between 0.14 and 0.28 (=about 0.21)
Chlorine	Fathead minnow	0.16
	Gammarus	0.16
Sulfides	Fathead minnow and white sucker (Catostomus commersoni)	0.1±
	Walleye pike (Stizostedion vitreum v.)	0.22±
Copper	Several species of fish	close to 0.1
Trivalent chromium	Fathead minnow	0.037
Hexavalent chromium	Fathead minnow	0.03
	Brook trout (Salvelinus fontinalis)	0.012
	Rainbow trout (Salmo gairdneri)	0.04
Malathion	Fathead minnow and bluegill (Lepomis machrochirus)	0.03
Carbaryl	Fish species	0.02
Nickel	Fathead minnow	0.02
Lead	Rainbow and Brook trout	<0.02
Zinc	Fathead minnow	0.005

In this approach, a 96-hour LC50 is determined for the pollutant using water from the receiving stream for dilution. The test organisms selected should be among the most sensitive species, or an important local species at a sensitive life stage, or a species whose relative sensitivity is known. This procedure takes into consideration the effects of local water quality and the stress or adverse effects of wastes already present in the stream. The LC50 thus found is then multiplied by the application factor for that waste to determine its safe concentration in the specific stream or section of stream. Such bioassays should be repeated at least monthly or when changes in process or rate of waste discharge are observed.

For example, if the 96-hour LC50 is 0.5 milligrams per liter (mg/l) and the concentration of the waste found to be safe is 0.01 mg/l, the ratio would be:

$$\frac{\text{Safe Concentration}}{\text{96-hour LC50}} = \frac{0.01}{0.50} = \frac{1}{50}$$

In this instance, the safe-to-lethal ratio is 0.02. It can be used as an application factor in other situations. Then, in a given situation involving this waste, the safe concentration in the receiving stream would be found by multiplying the four-day LC50 by 0.02.

This predictive procedure based on lethal concentrations is useful, because the precise safe level of many pollutants is not known because of the uncertainty about toxicity of mixed effluents and the difference in sensitivity among fish and fish food organisms. Henderson (1957)[27] and Tarzwell (1962)[49] have discussed various factors involved in developing application factors. Studies by Mount and Stephan (1967),[38] Brungs (1969),[18] Mount (1968),[37] McKim and Benoit (1971),[36] and Eaton (1970)[24] in which continuous exposure was used, reveal that the safe-to-lethal ratio that permits spawning ranges over nearly two orders of magnitude. Exposure will not be constant in most cases, and higher concentrations usually can be tolerated for short periods.

Lethal threshold concentrations, which may require more than 96-hour exposures, may be beneficially used (Sprague 1969)[43] to replace 96-hour LC50 in the above procedures, and there is a trend today to use such threshold concentrations (Eaton 1970).[24]

At present, safe levels have been determined for only a few wastes, and as a result only a few application factors are known. Because the determination of safe levels of pollutants is an involved process, interim procedures for estimating tolerable concentrations of various wastes in receiving waters must be used. To meet this situation, three universal application factors selected on the basis of present knowledge, experience, and judgment are recommended at the end of this section. Where toxicants have a nonpersistent nature (a half life of less than 4 days) or noncumulative effects, an application factor of 0.1 of the 96-hour LC50 should not be exceeded at any time or place after mixing with the receiving waters. The 24-hour average of the concentration of these toxicants should not exceed 0.05 of the LC50 if aquatic life is to be protected. For toxic materials which are persistent or cumulative the concentrations should not exceed 0.05 of the 96-hour LC50 at any time or place, and the 24-hour average concentration should not exceed 0.01 of the 96-hour LC50 in order to protect aquatic life. It is proposed that these general application factors be applied to LC50 values determined in the manner described above to set tolerable concentrations of wastes in the receiving stream.

MIXTURES OF TWO OR MORE TOXICANTS

The toxicity of a mixture of pollutants may be estimated by expressing the actual concentration of each toxicant as a proportion of its lethal threshold concentration (usually equal to the 96-hour LC50) and adding the resulting numbers for all the toxicants. If the total is 1.0 or greater, the mixture will be lethal.

The system of adding different toxicants in this way is based on the premise that their lethal actions are additive. Unlikely as it seems, this simple rule has been found to govern the combined lethal action of many pairs and mixtures of quite dissimilar toxicants, such as copper and ammonia, and zinc and phenol in the laboratory (Herbert and Vandyke 1964,[29] Jordan and Lloyd 1964,[30] Brown et al. 1969).[17] The rule holds true in field studies (Herbert 1965,[28] Sprague et al. 1965).[46] The method of addition is useful and reasonably accurate for predicting thresholds of lethal effects in mixtures.

There is also evidence of a lower limit for additive lethal effects. For ammonia and certain other pollutants, levels below 0.1 of the lethal concentration do not seem to contribute to the lethal action of a mixture (Brown et al. 1969,[17] Lloyd and Orr 1969).[35] This lower cutoff point of 0.1 of the LC50 should be used when it is necessary to assess the lethal effects of a mixture of toxicants.

SUBLETHAL EFFECTS

Sublethal or chronic effects of mixtures are of great importance. Sublethal concentrations of different toxicants should be additive in effect. Here again, it would be expected that for any given toxicant there would be some low concentration that would have no deleterious effect on an organism and would not contribute any sublethal toxicity to a mixture, but there is little research on this subject. Biesinger and Christensen (*unpublished data* 1971),[52] concluded that subchronic concentrations of 21 toxicants were close to being additive in causing chronic effects on reproduction in *Daphnia*. Copper and zinc concentrations of about 0.01 of the LC50 are additive in causing avoidance reactions (Sprague et al. 1965).[46] On the other hand, somewhat lower metal concentrations of about 0.003 of the LC50 do not seem to be additive in affecting reproduction of fish

(Eaton *unpublished data* 1971).[53] Perhaps there is a lower cutoff point than 0.01 of the LC50 for single pollutants contributing to sublethal toxicity of a mixture.

As an interim solution, it is recommended that the contribution of a single pollutant to the sublethal toxicity of a mixture should not be counted if it is less than 0.2 of the recommended level for that pollutant. Applying this to a basic recommended level of 0.05 (see the Recommendation that follows) of the LC50 would yield a value of 0.01 of the LC50, corresponding to the possible cutoff point suggested above.

It is expected that certain cases of joint toxicity will not be covered by simple addition. The most obvious exception would be when two toxicants combine chemically. For example, mixed solutions of cyanides and metals could cause addition of toxicity or very different effects if the metal and cyanide combined (Doudoroff et al. 1966).[23] A thorough understanding of chemical reactions is necessary in these cases.

For further discussions of bioassays and the difficulties posed in assessing sublethal effects of toxicants on organisms, see Section IV, pp. 233–237.

Recommendations for the Use of Application Factors to Estimate Safe Concentrations of Toxic Wastes in Receiving Streams

Where specific application factors have been determined for a given material, they should be used instead of the safe concentration levels of wastes given below:

(a) Concentration of materials that are nonpersistent or have noncumulative effects should not exceed 0.1 of the 96-hour LC50 at any time or place after mixing with the receiving waters. The 24-hour average of the concentration of these materials should not exceed 0.05 of the LC50 after mixing.

(b) For toxicants which are persistent or cumulative, the concentrations should not exceed 0.05 of the 96-hour LC50 at any time or place, nor should the 24-hour average concentration exceed 0.01 of the 96-hour LC50.

(c) When two or more toxic materials are present at the same time in the receiving water, it should be assumed unless proven otherwise that their individual toxicities are additive and that some reduction in the permissible concentrations is necessary. The amount of reduction required is a function of both the number of toxic materials present and their concentrations in respect to the permissible concentrations. The following relationship will assure that the combined amounts of the several substances do not exceed a permissible concentration:

$$\frac{C_a}{L_a}+\frac{C_b}{L_b}+\cdots+\frac{C_n}{L_n}\leq 1.0$$

This formula may be applied where C_a, C_b, . . . C_n are the measured or expected concentrations of the several toxic materials in the water, and L_a, L_b, . . . L_n are the respective concentrations recommended or those derived by using recommended application factors on bioassays done under local conditions. Should the sum of the several fractions exceed 1.0, a local restriction on the concentration of one or more of the substances is necessary.

C and L can be measured in any convenient chemical unit as proportions of the LC50 or in any other desired way, as long as the numerator and denominator of any single fraction are in the same units. To remove natural trace concentrations and low nonadditive concentrations from the above formula, any single fraction which has a value less than 0.2 should be removed from the calculation.

Example:

Small quantities of five toxicants are measured in a stream as follows:

3 micrograms/liter (μg/l) of zinc; 3 μg/l of phenol; 3 μg/l of un-ionized ammonia as calculated from Figure III-10 (see Ammonia, p. 186); 1 μg/l of cyanide; and 1 μg/l of chlorine.

A bioassay with zinc sulphate indicates that the 96-hour LC50 is 1.2 mg/l. The application factor for zinc is 0.005; therefore, the allowable limit is 0.005× 1.2 = 0.006 mg/l. Initial bioassays with phenol, ammonia, and cyanide indicate that the recommended values are the safe concentrations stated in other sections of the Report, not the fractions of LC50; so the limits are 0.1 mg/l, 0.02 mg/l, and 0.005 mg/l. The permissible limit for chlorine (page 189) is 0.003 mg/l. Therefore, the total toxicity is estimated as follows for zinc, phenol, ammonia, cyanide, and chlorine, respectively:

$$\frac{0.003}{0.006}+\frac{0.003}{0.1}+\frac{0.003}{0.02}+\frac{0.001}{0.005}+\frac{0.001}{0.003}$$

$$=0.5+0.03+0.15+0.2+0.33$$

The second and third terms, i.e., phenol and ammonia, should be deleted since they are below the minimum of 0.2 for additive effects. This leaves $0.5+0.2+0.33=1.03$, indicating that the total sublethal effect of these three toxicants is slightly above the permissible level and that no higher concentration of any of the three is safe. Thus none can be added as a pollutant.

PHYSICAL MANIPULATION OF THE ENVIRONMENT

Numerous activities initiated to maximize certain uses of water resources often adversely affect water quality and minimize other uses. These activities have caused both benefit and harm in terms of environmental quality. The common forms of physical alteration of watersheds are channelization, dredging, filling, shoreline modifications (of lakes and streams), clearing of vegetation, rip-rapping, diking, leveling, sand and gravel removal, and impounding of streams.

Channelization is widespread throughout the United States, and many studies have been conducted documenting its effects. Channelization usually increases stream gradient and flow rates. The quiet areas or backwaters are either eliminated or cut off from the main flow of the stream, the stream bed is made smooth, thus reducing the habitats available to benthic organisms, and surrounding marshes and swamps are more rapidly drained. The steeper gradient increases velocity allowing the stream to carry a greater suspended load and causing increased turbidity. The rate of organic waste transformation per mile is usually reduced, and destruction of spawning and nursery areas often occurs. Trautman (1939),[67] Smith and Larimore (1963),[65] Peters and Alvord (1964),[64] Welker (1967),[69] Martin (1969),[63] and Gebhards (1970)[58] have discussed the harmful effects of channelization on some fish populations and the effect on stimulation of less desirable species.

Dredging undertaken to increase water depth often destroys highly productive habitats such as marshes (Marshall 1968,[62] Copeland and Dickens 1969).[56] The spoils from dredging activities are frequently disposed of in other shallow sites causing further loss of productive areas. For example, Taylor and Saloman (1968)[66] reported that since 1950 there has been a 20 per cent decrease in surface area of productive Boca Ciega Bay, Florida, due to fill areas. It has become common practice to fill in marshy sites near large metropolitan areas (e.g., San Francisco Bay, Jamaica Bay) to provide for airport construction and industrial development.

In addition to the material that is actually removed by the dredging process, a considerable amount of waste is suspended in the water resulting in high turbidities (Mackin 1961).[61] If the dredged sediments are relatively nontoxic, gross effects on motile aquatic life may not be noticeable, but benthic communities may be drastically affected by the increased redeposition of silt (Ingle 1952).[59]

In many instances either high nutrient or toxic sediments are suspended or deposited during the dredging process. This action may kill aquatic organisms by exposure to the toxicants present or by the depletion of dissolved oxygen concentrations, or both. Brown and Clark (1968)[54] noted a dissolved oxygen reduction of 16 to 83 per cent when oxidizable sediments were resuspended. In many cases disturbed sediments containing high nutrient concentrations may stimulate undesirable forms of phytoplankton or *Cladophora*. Gannon and Beeton (1969)[57] categorized harbor sediments in five groups. Those most severely polluted were toxic to various animals and did not stimulate growth of phytoplankton. Other sediments were toxic but stimulated plant growth. The least polluted sediments were not toxic and stimulated growth of phytoplankton but not *Cladophora*.

Three basic aspects must be considered in evaluating the impact of dredging and disposal on the aquatic environment: (1) the amount and nature of the dredgings, (2) the nature and quality of the environments of removal and disposal, and (3) the ecological responses. All vary widely in different environments, and it is not possible to identify an optimal dredging and disposal system. Consequently, the most suitable program must be developed for each situation. Even in situations where soil is deposited in diked enclosures or used for fill, care must be taken to monitor overflow, seepage, and runoff waters for toxic and stimulatory materials.

Artificial impoundments may have serious environmental impact on natural aquatic ecosystems. Dams and other artificial barriers frequently block migration and may destroy large areas of specialized habitat. Aquatic organisms are frequently subjected to physical damage if they are allowed to pass through or over hydroelectric power units and other man-made objects when properly designed barriers are not provided. At large dams, especially those designed for hydroelectric power, water drawn from the pool behind the dam is frequently taken from great depths, resulting in the release to the receiving stream of waters low in dissolved oxygen and excessively cold. This can be a problem, particularly in areas where nonnative fish are stocked.

Cutting down forests, planting the land in crops, and partially covering the surface of a watershed by building roads, houses, and industries can have detrimental effects on water ways. Wark and Keller (1963)[66] showed that in the Potomac River Basin (Washington, D.C.) reducing the forest cover from 80 per cent to 20 per cent increased the annual sediment yield from 50 to 400 tons per square mile per year. The planting of land in crops increased the sediment yield from 70 to 300 tons per square mile per year, or a fourfold increase as the land crops increased from 10 per cent to 50 per cent. Likens et al. (1970)[60] showed that cutting down the forest in the Hubbard Brook area (Vermont) caused substantial changes in the streams. The sediment load increased fourfold over a period from May 1966 to May 1968. Furthermore, the particulate matter drained from the deforested watershed became increasingly inorganic in content, thus reducing the value of the sediment as a food source. The nutrient content of the water was also affected by cutting down the forests. The nitrate concentration increased from 0.9 mg/l prior to the cutting of vegetation to 53 mg/l two years later. Temperatures of streams in deforested areas were higher, particularly during the summer months, than those of streams bordered by forests (Brown and Krygier 1970).[55]

Prior to any physical alterations of a watershed, a thorough investigation should be conducted to determine the expected balance between benefits and adverse environmental effects.

SUSPENDED AND SETTLEABLE SOLIDS

Suspended and settleable solids include both inorganic and organic materials. Inorganic components include sand, silt, and clay originating from erosion, mining, agriculture, and areas of construction. Organic matter may be composed of a variety of materials added to the ecosystem from natural and man-made sources. These inorganic and organic sources are discussed in the Panel Report on Marine Aquatic Life and Wildlife (Section IV), and the effects of land–water relationships are described in the report on Recreation and Aesthetics (Section I).

SOIL AS A SOURCE OF MINERAL PARTICLES

Soil structure and drainage patterns, together with the intensity and temporal distribution of rainfall that directly affect the kind and amount of protective vegetative cover, determine the susceptibility of a soil to erosion. Where rain occurs more or less uniformly throughout the year, protective grasses, shrubs, or trees develop (Leopold, et al. 1964).[78] Where rainfall occurs intermittently, as in arid areas, growth of protective plants is limited thus allowing unchecked erosion of soils.

Wetting and drying cause swelling and shrinking of clay soils and leave the surface susceptible to entrainment in surface water flows. Suspended soil particle concentrations in rivers, therefore, are at their peak at the beginning of flood flows. Data on the concentration of suspended matter in most of the significant streams of the United States are presented in the U.S. Geological Survey Water Supply Papers.

Streams transport boulders, rocks, pebbles, and sand by intermittent rolling motions, or by intermittent suspension and deposition as particles are entrained and later settled on the bed. Fine particles are held in suspension for long periods, depending on the intensity of the turbulence. Fine silt particles, when dispersed in fresh waters, remain almost continuously suspended, and suspension of dispersed clay mineral particles may be maintained even by the thermally induced motions in water. These fine mineral particles are the soil materials of greatest significance to the turbidity values of a particular water.

The suspended and settleable solids and the bed of a water body must be considered as interrelated, interacting parts. For example, Langlois (1941)[77] reported that in Lake Erie the average of 40 parts per million (ppm) of suspended matter in the water was found to change quickly to more than 200 ppm with a strong wind. He further explained that this increase is attributed to sediments resuspended by wave action. These sediments enter from streams or from shoreline erosion.

Suspended clay mineral particles are weakly cohesive in fresh river waters having either unusually low dissolved salt concentrations or high concentrations of multivalent cations. Aggregations of fine particles form and settle on the bed to form soft fluffy deposits when such waters enter a lake or impoundment. However, clay mineral particles are dispersed or only weakly cohesive in most rivers.

EFFECTS OF SUSPENDED PARTICLES IN WATER

The composition and concentrations of suspended particles in surface waters are important because of their effects on light penetration, temperature, solubility products, and aquatic life (Cairns 1968).[72] The mechanical or abrasive action of particulate material is of importance to the higher aquatic organisms, such as mussels and fish. Gills may be clogged and their proper functions of respiration and excretion impaired. Blanketing of plants and sessile animals with sediment as well as the blanketing of important habitats, such as spawning sites, can cause drastic changes in aquatic ecosystems. If sedimentation, even of inert particles, covers substantial amounts of organic material, anaerobic conditions can occur and produce noxious gases and other objectionable characteristics, such as low dissolved oxygen and decreases in pH.

Absorption of sunlight by natural waters is strongly affected by the presence of suspended solids. The intensity of light (I) at any distance along a light ray (L) is, for a uniform suspension, expressed by the formula:

$$I = I_0{}^{-kcL},$$

where I_0 is the intensity just below the water surface ($L = 0$),

k is the extinction coefficient for the suspended solids, and c is the concentration of suspended solids. L can be related to the water depth by the zenith angle, i, the angle of refraction, r, and the index of refraction of water, 1.33, by Snell's rule:

$$\sin r = \frac{\sin i}{1.33}$$

The depth, D, is $L \cos r$. Refraction makes the light path more nearly vertical under water than the sun's rays, except when the sun's rays are themselves normal to the water surface.

The growth of fixed and suspended aquatic plants can be limited by the intensity of sunlight. An example of the decrease in the photic zone was calculated for San Francisco Bay (Krone 1963),[76] where k was 1.18×10^3 square centimeters per gram. For a typical suspended solids concentration of 50×10^{-6} grams per cubic centimeter, for an algae requiring 20 foot candles or more for its multiplication, and under incident sunlight of 13,000 foot candles the photic zone did not exceed 1.1 meters. A reduction in suspended solids concentration to 20×10^{-6} g/cm^3 increased the maximum depth of the photic zone to 2.8 meters.

Because suspended particles inhibit the penetration of sunlight, water temperatures are affected, and increasing turbidity results in increasing absorption near the water surface so that turbid waters warm more rapidly at the surface than do clearer waters. Warming and the accompanying decrease in density stabilize water and may inhibit vertical mixing. Lower oxygen transfer value from air to water results when surface waters are heated. This action combined with inhibited vertical mixing reduces the rate of oxygen transfer downward. Still or slowly moving water is most affected.

The rate of warming, dT/dt, at any distance from the surface along a light path, L, in water having uniform suspended material is

$$\frac{dT}{dt} = -\left[\frac{Ikc}{\rho C}\right]^{-kcL}$$

where ρ is the water density and C is the specific heat of the water. This equation shows that an increase in suspended sediment concentration increases the rate of warming near the surface and decreases exponentially with depth. The biological significance of this relationship is in the effect on time of formation, vertical distribution of thermal stratification, and stability of the upper strata. Increasing turbidity could change the stratification patterns of a lake and thus change the temperature distribution, oxygen regime, and composition of the biological communities.

ADSORPTION OF TOXIC MATERIALS

Suspended mineral particles have irregular, large surface areas, with electrostatic charges. As a consequence, clay minerals may sorb cations, anions, and organic compounds. Pesticides and heavy metals may be absorbed on suspended clay particles and strongly held with them. The sorption of chemicals by suspended matter is particularly important if it leads to a buildup of toxic and radioactive materials in a limited area with the possibility of sudden release of these toxicants. One such example has been reported by Benoit et al. (1967).[70] Gannon and Beeton (1969)[75] reported that sediments with the following characteristics dredged from various harbors on the Great Lakes were usually toxic to various organisms: COD 42,000 mg/l, volatile solids 4,000 mg/l, ammonia 0.075 mg/g, phosphate-P 0.65 mg/g.

The capacity of minerals to hold dissolved toxic materials is different for each material and type of clay mineral. An example illustrates the magnitudes of sorptive capacities: the cation exchange capacity (determined by the number of negatively charged sites on clay mineral surfaces) ranges from a few milliequivalents per hundred grams (me/100 g) of mineral for kaolinite clay to more than 100 me/100 g for montmorillonite clay. Typical estuarial sediments, which are mixtures of clay, silt, and sand minerals, have exchange capacities ranging from 15 to 60 me/100 g (Krone 1963).[76] The large amounts of such material that enter many estuaries and lakes from tributary streams provide continually renewed sorptive capacity that removes materials such as heavy metals, phosphorus, and radioactive ions. The average new sediment load flowing through the San Francisco Bay-Delta system, for example, has a total cation exchange capacity of a billion equivalents per year.

The sorptive capacity effectively creates the large assimilative capacity of muddy waters. A reduction in suspended mineral solids in surface waters can cause an increase in the concentrations of dissolved toxic materials contributed by existing waste discharges.

EFFECTS ON FISH AND INVERTEBRATES

The surface of particulate matter may act as a substratum for microbial species, although the particle itself may or may not contribute to their nutrition. When the presence of particulate matter enables the environment to support substantial increased populations of aquatic microorganisms, the dissolved oxygen concentration, pH, and other characteristics of the water are frequently altered.

There are several ways in which an excessive concentration of finely divided solid matter might be harmful to a fishery in a river or a lake (European Inland Fisheries Advisory Commission, EIFAC 1965).[73] These include:

- acting directly on fish swimming in water in which solids are suspended, either killing them or reducing their growth rate and resistance to disease;
- preventing the successful development of fish eggs and larvae;

- modifying natural movements and migrations of fish;
- reducing the food available to fish;
- affecting efficiency in catching the fish.

With respect to chemically inert suspended solids and to waters that are otherwise satisfactory for the maintenance of freshwater fisheries, EIFAC (1965)[73] reported:

- there is no evidence that concentrations of suspended solids less than 25 mg/l have any harmful effects on fisheries;
- it should usually be possible to maintain good or moderate fisheries in waters that normally contain 25 to 80 mg/l suspended solids; other factors being equal, however, the yield of fish from such waters might be somewhat lower than from those in the preceding category;
- waters normally containing from 80 to 400 mg/l suspended solids are unlikely to support good freshwater fisheries, although fisheries may sometimes be found at the lower concentrations within this range;
- only poor fisheries are likely to be found in waters that normally contain more than 400 mg/l suspended solids.

In addition, although several thousand parts per million suspended solids may not kill fish during several hours or days exposure, temporary high concentrations should be prevented in rivers where good fisheries are to be maintained. The spawning grounds of most fish should be kept as free as possible from finely divided solids.

While the low turbidities reported above reflected values that should protect the ecosystem, Wallen (1951)[80] reported that fish can tolerate higher concentrations. Behavioral reactions were not observed until concentrations of turbidity neared 20,000 mg/l, and in one species reactions did not appear until turbidities reached 100,000 mg/l. Most species tested endured exposures of more than 100,000 mg/l turbidity for a week or longer, but these same fishes finally died at turbidities of 175,000 to 225,000 mg/l. Lethal turbidities caused the death of fishes within 15 minutes to two hours exposure. Fishes that succumbed had opercular cavities and gill filaments clogged with silty clay particles from the water.

In a study of fish and macroinvertebrate populations over a four-year period in a stream receiving sediment from a crushed limestone quarry, Gammon (1970)[74] found that inputs that increased the suspended solids load less than 40 mg/l (normal suspended solids was 38 to 41 mg/l and volatile suspended solids 16 to 30 mg/l) resulted in a 25 per cent reduction in macroinvertebrate density in the stream below the quarry. A heavy silt input caused increases of more than 120 mg/l including some decomposition of sediment, and resulted in a 60 per cent reduction in density of macroinvertebrates. Population diversity indices were unaffected because most species responded to the same degree. The standing crop of fish decreased dramatically when heavy sediment occurred in the spring; but fish remained in pools during the summer when the input was heavy and vacated the pools only after deposits of sediment accumulated. After winter floods removed sediment deposits, fish returned to the pools and achieved levels of 50 per cent of the normal standing crop by early June.

Not all particulate matter affects organisms in the same way. For example, Smith, et al. (1965)[79] found that the lethal action of pulp-mill fiber on walleye fingerlings (*Stizostedion vitreum vitreum*) and fathead minnows (*Pimephales promelas*) was influenced by the type of fiber. In 96-hour bioassays, mortality of the minnows in 2,000 ppm suspensions was 78 per cent in conifer groundwood, 34 per cent in conifer kraft, and 4 per cent in aspen groundwood. High temperatures and reduced dissolved oxygen concentrations increased the lethal action of fiber.

Buck (1956)[71] studied the growth of fish in 39 farm ponds having a wide range of turbidities. The ponds were cleared of fish and then restocked with largemouth black bass (*Micropterus salmoides*), bluegill (*Lepomis macrochirus*), and redear sunfish (*Lepomis microlophus*). After two growing seasons the yields of fish were:

• clear ponds (less than 25 mg/l suspended solids)	161.5 lb/acre
• intermediate (25–100 mg/l suspended solids)	94.0 lb/acre
• muddy (more than 100 mg/l suspended solids)	29.3 lb/acre

The rate of reproduction was also reduced by turbidity, and the critical concentration for all three species appeared to be about 75–100 mg/l. In the same paper, Buck reported that largemouth black bass (*Micropterus salmoides*), crappies (*Pomoxis*), and channel catfish (*Ictalurus punctatus*) grew more slowly in a reservoir where the water had an average turbidity of 130 mg/l than in another reservoir where the water was always clear.

Floating materials, including large objects as well as very fine substances, can adversely affect the activities of aquatic life. Floating logs shut out sunlight and interfere particularly with surface feeding fish. Logs may also leach various types of organic acids due to the action of water. If they have been sprayed with pesticides or treated chemically, these substances may also leach into the water. As the logs float downstream their bark often disengages and falls to the bed of the stream, disturbing benthic habitats. Aquatic life is also affected by fine substances, such as sawdust, peelings, hair from tanneries, wood fibers, containers, scum, oil, garbage, and materials from untreated municipal and industrial wastes, tars and greases, and precipitated chemicals.

Recommendations

- **The combined effect of color and turbidity should not change the compensation point more than 10 per cent from its seasonally established norm, nor should such a change place more than 10 per cent of the biomass of photosynthetic organisms below the compensation point.**

- **Aquatic communities should be protected if the following maximum concentrations of suspended solids exist:**

High level of protection	**25 mg/l**
Moderate protection	**80 mg/l**
Low level of protection	**400 mg/l**
Very low level of protection	**over 400 mg/l**

COLOR

The true color of a specific water sample is the result of substances in solution; thus it can be measured only after suspended material has been removed. Color may be of organic or mineral origin and may be the result of natural processes as well as manufacturing operations. Organic sources include humic materials, peat, plankton, aquatic plants, and tannins. Inorganic substances are largely metallic, although iron and manganese, the most important substances, are usually not in solution. They affect color as particles. Heavy-metal complexes are frequent contributors to the color problem.

Many industries (such as pulp and paper, textile, refining, chemicals, dyes and explosives, and tanning) discharge materials that contribute to the color of water. Conventional biological waste treatment procedures are frequently ineffective in removing color. On the other hand, such treatment processes have caused an accentuation of the level of color during passage through the treatment plant. Physicochemical treatment processes are frequently preferable to biological treatment if color removal is critical (Eye and Aldous 1968,[81] King and Randall 1970[83]).

The tendency for an accentuation of color to occur as a result of complexing of a heavy metal with an organic substance may also lead to problems in surface waters. A relatively color-free discharge from a manufacturing operation, may, upon contact with iron in a stream, produce a highly colored water that would significantly affect aquatic life (Hem 1960,[82] Stumm and Morgan 1962[86]).

The standard platinum-cobalt method of measuring color is applicable to a wide variety of water samples (Standard Methods 1971).[85] However, industrial wastes frequently produce colors dissimilar to the standard platinum-cobalt color, making the comparison technique of limited value. The standard unit of color in water is that level produced by 1 mg/l of platinum as chloroplatinate ion (Standard Methods 1971).[85] Natural color in surface waters ranges from less than one color unit to more than 200 in highly colored bodies of water (Nordell 1961).[84]

That light intensity at which oxygen production in photosynthesis and oxygen consumption by respiration of the plants concerned are equal is known as the compensation point, and the depth at which the compensation point occurs is called the compensation depth. For a given body of water this depth varies with several conditions, including season, time of day, the extent of cloud cover, condition of the water, and the taxonomic composition of the flora involved. As commonly used, the compensation point refers to that intensity of light which is such that the plant's oxygen production during the day will be sufficient to balance the oxygen consumption during the whole 24-hour period (Welch 1952).[87]

Recommendation

The combined effect of color and turbidity should not change the compensation point more than 10 per cent from its seasonally established norm, nor should such a change place more than 10 per cent of the biomass of photosynthetic organisms below the compensation point.

DISSOLVED GASES

DISSOLVED OXYGEN

Oxygen requirements of aquatic life have been extensively studied. Comprehensive papers have been presented by Doudoroff and Shumway (1967),[89] Doudoroff and Warren (1965),[91] Ellis (1937),[92] and Fry (1960).[94] (Much of the research on temperature requirements also considers oxygen, and references cited in the discussion of Heat and Temperature, p. 151, are relevant here.) The most comprehensive review yet to appear has been written by Doudoroff and Shumway for the Food and Agriculture Organization (FAO) of the United Nations (1970).[90] This FAO report provides the most advanced summary of scientific research on oxygen needs of fish, and it has served as a basis for most of the recommendations presented in this discussion. In particular, it provided the criteria for citing different levels of protection for fish, for change from natural levels of oxygen concentration, and for the actual numerical values recommended. Much of the text below has been quoted verbatim or condensed from the FAO report. Its recommendations have been modified in only two ways: the insertion of a floor of 4 mg/l as a minimum, and the suggestion that natural minima be assumed to be equal to saturation levels if the occurrence of lower minima cannot be definitely established. Doudoroff and Shumway covered oxygen concentrations below the floor of 4 mg/l; however, the 4 mg/l floor has been adopted in this report for reasons explained below.

Levels of Protection

Most species of adult fish can survive at very low concentrations of dissolved oxygen. Even brook trout (*Salvelinus fontinalis*) have been acclimated in the laboratory to less than 2 mg/l of O_2. In natural waters, the minimum concentration that allows continued existence of a varied fish fauna, including valuable food and game species, is not high. This minimum is not above 4 mg/l and may be much lower.

However, in evaluating criteria, it is not important to know how long an animal can resist death by asphyxiation at low dissolved oxygen concentrations. Instead, data on the oxygen requirements for egg development, for newly hatched larvae, for normal growth and activity, and for completing all stages of the reproductive cycle are pertinent. Upon review of the available research, one fact becomes clear: *any* reduction of dissolved oxygen can reduce the efficiency of oxygen uptake by aquatic animals and hence reduce their ability to meet demands of their environment. There is evidently no concentration level or percentage of saturation to which the O_2 content of natural waters can be reduced without causing or risking some adverse effects on the reproduction, growth, and consequently, the production of fishes inhabiting those waters.

Accordingly, no single, arbitrary recommendation can be set for dissolved oxygen concentrations that will be favorable for all kinds of fish in all kinds of waters, or even one kind of fish in a single kind of water. Any reduction in oxygen may be harmful by affecting fish production and the potential yield of a fishery.

The selection of a level of protection (Table III–3) is a socioeconomic decision, not a biological one. Once the level of protection is selected, appropriate scientific recommendations may be derived from the criteria presented in this discussion.

Basis for Recommendations

The decision to base the recommendations on O_2 concentration minima, and not on average concentrations, arises from various considerations. Deleterious effects on fish seem to depend more on extremes than on averages. For example, the growth of young fish is slowed markedly if the oxygen concentration falls to 3 mg/l for part of the day, even if it rises as high as 18 mg/l at other times. It could be an inaccurate and possibly controversial task to carry out the sets of measurements required to decide whether a criterion based on averages was being met.

A daily fluctuation of O_2 is to be expected where there is appreciable photosynthetic activity of aquatic plants. In such cases, the minimum O_2 concentration will usually be found just before daybreak, and sampling should be done at that time. Sampling should also take into account the possible differences in depth or width of the water body. The guiding principle should be to sample the places where aquatic organisms actually live or the parts of the habitat where they should be able to live.

Before recommendations are proposed, it is necessary to evaluate criteria for the natural, seasonal O_2 minimum from which the recommendations can be derived. Natural levels are assumed to be the saturation levels, unless scientific data show that the natural levels were already low in the absence of man-made effects.

Certain waters in regions of low human populations can still be adequately studied in their natural or pristine condition. In these cases the minimum O_2 concentration at different seasons, temperatures, and stream discharge volumes can be determined by direct observation. Such observed conditions can also be useful in estimating seasonal minima in similar waters in similar geographical regions where natural levels can no longer be observed because of waste discharges or other man-made changes.

In many populated regions, some or all of the streams and lakes have been altered. Direct determination of natural minima may no longer be possible. In these cases the assumption of year-round saturation with O_2 is made in the absence of other evidence.

Supersaturation of water with dissolved oxygen may occur as the result of photosynthesis by aquatic vegetation. There is some evidence that this may be deleterious to aquatic animals because of gas bubble disease (see Total Dissolved Gases, p. 135).

Despite the statements in previous paragraphs that there is no single O_2 concentration which is *favorable* to all species and ecosystems, it is obvious that there are, nevertheless, very low O_2 concentrations that are *unfavorable* to almost all aquatic organisms. Therefore, a floor of 4 mg/l is recommended except in situations where the natural level of dissolved oxygen is less than 4 mg/l in which case no further depression is desirable. The value of 4 mg/l has been selected because there is evidence of subacute or chronic damage to several fish below this concentration. Doudoroff and Shumway (1970)[90] review the work of several authors as given below, illustrating such damage. Fathead minnows (*Pimephales promelas*) held at 4 mg/l spawned satisfactorily; only 25 per cent of the resultant fry survived for 30 days, compared to 66 per cent survival at 5 mg/l. At an oxygen level of 3 mg/l, survival of fry was even further reduced to 5 per cent (Brungs 1972[101] *personal communication*). Shumway et al. (1964)[98] found that the dry weight of coho salmon (*Oncorhynchus kisutch*) alevins (with yolk sac removed) was reduced by 59 per cent when they had been held at 3.8 mg/l of oxygen, compared to weights of the controls. The embryos of sturgeon (*Acipenser*) suffered complete mortality at oxygen concentrations of 3.0 to 3.5 mg/l, compared to only 18 per cent mortality at 5.0 to 5.5 mg/l (Yurovitskii 1964).[100] Largemouth bass (*Micropterus salmoides*) embryos reared at 25 C showed survival equal to controls only at oxygen levels above 3.5 mg/l (Dudley 1969).[92] Efficiency of food conversion by juvenile bass was nearly independent of O_2 at 5 mg/l and higher, but growth rate was reduced by 16.5 per cent at 4 mg/l, and 30 per cent at 3 mg/l (Stewart et al. 1967).[99] Similar reductions in growth of underyearling coho salmon occurred at the same O_2 concentrations (Herrmann et al. 1962).[95] Although many other experiments have shown little or no damage to performance of fish at 4 mg/l, or lower, the evidence given above shows appreciable effects on embryonic and juvenile survival and growth for several species of fish sufficient to justify this value.

Warm- and Coldwater Fishes

There are many associations and types of fish fauna throughout the country. Dissolved oxygen criteria for coldwater fishes and warmwater game fishes are considered together in this report. There is no evidence to suggest that the more sensitive warmwater species have lower O_2 requirements than the more sensitive coldwater fishes. The difference in O_2 requirements is probably not greater than the difference of the solubility of O_2 in water at the maximum temperatures to which these two kinds of fish are normally exposed in summer (Doudoroff and Shumway 1970).[90] In warmwater regions, however, the variety of fishes and fish habitats is relatively great, and there are many warmwater species that are exceedingly tolerant of O_2 deficiency.

Unusual Waters

There are certain types of waters that naturally have low oxygen content, such as the "black waters" draining swamps of the Southeastern United States. (Other examples include certain deep ocean waters and eutrophic waters that support heavy biomass, the respiration of which reduces O_2 content much of the time.) A special situation prevails in the deep layers (hypolimnion) of some lakes. Such layers do not mix with the surface layers for extended periods and may have reduced O_2, or almost none. Fish cannot live in the deep layers of many such lakes during a large part of the year, although each lake of this kind must be considered as a special case. However, the recommendation that *no* oxygen-consuming wastes should be released into the deep layers still applies, since there may be no opportunity for reaeration for an entire season.

Organisms Other Than Fish

Most research concerning oxygen requirements for freshwater organisms deals with fish; but since fish depend upon other aquatic species for food, it is necessary to consider the O_2 requirements of these organisms. This Section makes the assumption that the O_2 requirements of other components of the aquatic community are compatible with fish (Doudoroff and Shumway 1970).[90] There are certain exceptions where exceedingly important invertebrate organisms may be very sensitive to low O_2, more sensitive than the fish species in that habitat (Doudoroff and Shumway 1970).[90]

The situation is somewhat more complicated for invertebrates and aquatic plants, inasmuch as organic pollution that causes reduction of O_2 also directly increases food material. However, it appears equally true for sensitive invertebrates as for fish that *any* reduction of dissolved O_2 may have deleterious effects on their production. For example, Nebeker (1972)[97] has found that although a certain mayfly (*Ephemera simulans*) can survive at 4.0 mg/l of oxygen for four days, any reduction of oxygen below saturation causes a decrease in successful transformation of the immature to the adult stage.

Salmonid Spawning

For spawning of salmonid fishes during the season when eggs are in the gravel, there are even greater requirements for O_2 than those given by the high level of protection. (See Table III-3 for description of levels.) This is because the water associated with the gravel may contain less oxygen than the water in the stream above the gravel. There is abundant evidence that salmonid eggs are adversely affected in direct proportion to reduction in O_2. The oxygen criteria for eggs should be about half way between the nearly maximum and high levels of protection.

TABLE III-3—Guidelines for Selecting Desired Type and Level of Protection of Fish Against Deleterious Effects of Reduced Oxygen Concentrations

Level of protection	Intended type of protection	Possible application
Nearly Maximum[a]	For virtually unimpaired productivity and unchanged quality of a fishery.	Appropriate for conservation areas, parks, and water bodies of high or unique value. Requires, practically speaking, that little or no deoxygenating wastes be added to natural waters. Nor must there be any activities such as unfavorable land use which would reduce O_2 level.
High	Not likely to cause appreciable change in the ecosystem, nor material reduction of fish production. Some impairment is risked, but appreciable damage is not to be expected at these levels of oxygen.	Could be appropriate for fisheries or aquatic ecosystems of some importance, which should not be impaired by other uses of water.
Moderate	Fisheries should persist, usually with no serious impairment, but with some decrease in production.	Could be used for fisheries which are valued, but must co-exist with major industries or dense human population.
Low	Should permit the persistence of sizeable populations of tolerant species and successful passage of most migrants[b]. Much reduced production or elimination of sensitive fish is likely.	Appropriate for fisheries that have some commercial or recreational value, but are so unimportant compared with other water uses, that their maintenance cannot be a major objective of pollution control. This type of protection should however, provide for survival of sensitive species in adult or subadult life stages for short periods during the year, if oxygen levels at other times are satisfactory for growth, reproduction, etc.

[a] Note that there could be a higher level of protection that would require oxygen to be near natural level at all times, whereas nearly maximum requires only that oxygen should not fall below the lowest level characteristic of the season.

[b] But will not protect migrating salmonids, which would require at least a Moderate level of protection, for zones of passage.

Interaction with Toxic Pollutants or Other Environmental Factors

It is known that reduced oxygen levels increase the toxicity of pollutants. A method for predicting this interaction has been given by Brown (1968),[88] and a theoretical background by Lloyd (1961).[96] The disposal of toxic pollutants must be controlled so that their concentrations will not be unduly harmful at prescribed acceptable levels of O_2, temperature, and pH. The levels of oxygen recommended in this Section are independent of the presence of toxic wastes, no matter what the nature of the interaction between these toxicants and O_2 deficiencies. Carbon dioxide is an exception, because its concentration influences the safe level of oxygen. The recommendations for O_2 are valid when the CO_2 concentration is within the limits recommended in the section on CO_2.

Application of Recommendations

As previously stated, the recommendations herein differ in two important respects from those widely used. First, they are not fixed values independent of natural conditions. Second, they offer a choice of different levels of protection of fishes, the selection of any one of which is primarily a socioeconomic decision, not a biological one.

Table III-4 presents guidelines for the protection of fishes at each of four levels. Each column shows the level to which the dissolved O_2 can be reduced and still provide the stated level of protection for local fisheries. The values can be derived from the equations given in the recommendations. These equations have been calculated to fit the curves shown in the figure on page 264 of Doudoroff and Shumway (1970),[90] which serve as the basis of the recommendations. To use Table III-4, the estimated natural seasonal minimum should first be determined on the basis of available data or from expert judgment. This may be taken to be the minimum saturation value for the season, unless there is scientific evidence that losses of O_2 levels prevailed naturally. The word "season" here means a period based on local climatic and hydrologic conditions, during which the natural thermal and dissolved O_2 regime of a stream or lake can be expected to be fairly uniform. Division of the year into equal three-month periods, such as December–February, March–May, is satisfactory. However, under special conditions, the designated seasons could be periods longer or shorter than three months, and could in fact be taken as individual months. The selected periods need not be equal in length.

When the lowest natural value for the season has been estimated, the desired kind and level of protection should then be selected according to the guidelines in Table III-3. The recommended minimum level of dissolved oxygen may then be found in the selected column of Table III-4, or as given by the formula in the recommendation.

TABLE III-4—Example of Recommended Minimum Concentrations of Dissolved Oxygen

Estimated natural seasonal minimum concentration of oxygen in water	Corresponding temperature of oxygen-saturated fresh water	Recommended minimum concentrations of O_2 for selected levels of protection			
		Nearly maximal	High	Moderate	Low
5	(*a*) (*a*)	5	4.7	4.2	4.0
6	46C(a) (115F)(a)	6	5.6	4.8	4.0
7	36C (96.8F)	7	6.4	5.3	4.0
8	27.5C (81.5F)	8	7.1	5.8	4.3
9	21C (69.8F)	9	7.7	6.2	4.5
10	16C (60.8F)	10	8.2	6.5	4.6
12	7.7C (45.9F)	12	8.9	6.8	4.8
14	1.5C (34.7F)	14	9.3	6.8	4.9

[a] Included to cover waters that are naturally somewhat deficient in O_2. A saturation value of 5 mg/l might be found in warm springs or very saline waters. A saturation value of 6 mg/l would apply to warm sea water (32 C=90 F).

Note: The desired kind and level of protection of a given body of water should first be selected (across head of table). The estimated seasonal minimum concentration of dissolved oxygen under natural conditions should then be determined on the basis of available data, and located in the left hand column of the table. The recommended minimum concentration of oxygen for the season is then taken from the table. All values are in milligrams of O_2 per liter. Values for natural seasonal minima other than those listed are given by the formula and qualifications in the section on recommendations.

Examples

- It is desired to give moderate protection to trout (*Salvelinus fontinalis*) in a small stream during the summer. The maximum summer temperature is 20 C (68 F); the salt content of the water is low and has negligible effect on the oxygen saturation value. The atmospheric pressure is 760 millimeters (mm) Hg. Oxygen saturation is therefore 9.2 mg/l. This is assumed to be the natural seasonal minimum in the absence of evidence of lower natural concentrations. Interpolating from Table III-4 or using the recommended formula, reveals a minimum permissible concentration of oxygen during the summer of 6.2 mg/l. If a high level of protection had been selected, the recommendation would have been 7.8 mg/l. A low level of protection, providing little or no protection for trout but some for more tolerant fish, would require a recommendation of 4.5 mg/l. Other recommendations would be calculated in a similar way for other seasons.
- It is decided to give moderate protection to largemouth bass (*Micropterus salmoides*) during the summer. Stream temperature reaches a maximum of 35 C (95 F) during summer, and lowest seasonal saturation value is accordingly 7.1 mg/l. The recommendation for minimum oxygen concentration is 5.4 mg/l.
- For low protection of fish in summer in the same stream described above (for largemouth bass), the recommendation would be 4.0 mg/l, which is also the floor value recommended.
- It is desired to protect marine fish in full-strength sea water (35 parts per thousand salinity) with a maximum seasonal temperature of 16 C (61 F). The saturation value of 8 mg/l is assumed to be the natural dissolved oxygen minimum for the season. For a high level of protection, the recommendation is 7.1 mg/l, for a moderate level of protection it is 5.8 mg/l, and for a low level of protection it is 4.3 mg/l.

It should be stressed that the recommendations are the minimum values for any time during the same season.

Recommendations

(a) For nearly maximal protection of fish and other aquatic life, the minimum dissolved oxygen in any season (defined previously) should not be less than the estimated natural seasonal minimum concentration (defined previously) characteristic of that body of water for the same season. In estimating natural minima, it is assumed that waters are saturated, unless there is evidence that they were lower in the absence of man-made influences.

(b) For a high level of protection of fish, the minimum dissolved oxygen concentration in any season should not be less than that given by the following formula in which M = the estimated natural seasonal minimum concentration characteristic of that body of water for the same season, as qualified in (a):

$$\text{Criterion}^{*} = 1.41M - 0.0476M^2 - 1.11$$

(c) For a moderate level of protection of fish, the minimum dissolved oxygen concentration in any season should not be less than is given by the following formula with qualifications as in (b):

$$\text{Criterion}^{*} = 1.08M - 0.0415M^2 - 0.202$$

(d) For a low level of protection of fish, the minimum O_2 in any season should not be less than given by the following formula with qualifications as in (b):

$$\text{Criterion}^{*} = 0.674M - 0.0264M^2 + 0.577$$

(e) A floor value of 4 mg/l is recommended except in those situations where the natural level of dissolved oxygen is less than 4 mg/l, in which case no further depression is desirable.

(f) For spawning grounds of salmonid fishes, higher O_2 levels are required as given in the following formula with qualifications as in (b):

$$\text{Criterion}^{*} = 1.19M - 0.0242M^2 - 0.418$$

(g) In stratified eutrophic and dystrophic lakes, the dissolved oxygen requirements may not apply to the hypolimnion and such lakes should be considered on a case by case basis. In other stratified lakes, recommendations (a), (b), (c), and (d) apply; and if the oxygen is below 4 mg/l, recommendation (e) applies. In unstratified lakes recommendations apply to the entire circulating water mass.

(i) All the foregoing recommendations apply to all waters except waters designated as mixing zones

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[*] All values are instantaneous, and final value should be expressed to two significant figures.

(see section on Mixing Zones p. 112). In locations where supersaturation occurs, the increased levels of oxygen should conform to the recommendations in the discussion of Total Dissolved Gases, p. 139.

TOTAL DISSOLVED GASES (SUPERSATURATION)

Excessive total dissolved gas pressure (supersaturation) is a relatively new aspect of water quality. Previously, supersaturation was believed to be a problem that was limited to the water supplies of fish culture facilities (Shelford and Allee 1913).[135] Lindroth (1957)[126] reported that spillways at hydroelectric dams in Sweden caused supersaturation, and recently Ebel (1969)[112] and Beiningen and Ebel (1968)[103] established that spillways at dams caused gas bubble disease to be a limiting factor for aquatic life in the Columbia and Snake Rivers. Renfro (1963)[133] and others reported that excessive algal blooms have caused gas bubble disease in lentic water. DeMont and Miller (*in press*)[110] and Malous et al. (1972)[127] reported gas bubble disease among fish and mollusks living in the heated effluents of steam generating stations. Therefore, modified dissolved gas pressures as a result of dams, eutrophication, and thermal discharges present a widespread potential for adversely affecting fish and aquatic invertebrates. Gas bubble disease has been studied frequently since Gorham (1898,[119] 1899[120]) published his initial papers, with the result that general knowledge of the causes, consequences, and adverse levels are adequate to evaluate criteria for this water quality characteristic.

Gas bubble disease is caused by excessive total dissolved gas pressure *but it is not caused by the dissolved nitrogen gas alone* (Marsh and Gorham 1904,[128] Shelford and Allee 1913,[135] Englehorn 1943,[115] Harvey et al. 1944a,[121] Doudoroff 1957,[111] Harvey and Cooper 1962).[123] Englehorn (1943)[115] analyzed the gases contained in the bubbles that were formed in fish suffering from gas bubble disease and found that their gas composition was essentially identical to air. This was confirmed by Shirahata (1966).[136]

Etiologic Factors

Gas bubble disease (GBD) results when the uncompensated total gas pressure is greater in the water than in the air, but several important factors influence the etiology of GBD. These factors include: exposure time and physical factors such as hydrostatic pressure; other compensating forces and biological factors such as species or life stage tolerance or levels of activity; and any other factors that influence gas solubility. Of these factors perhaps none are more commonly misunderstood than the physical roles of total dissolved gas pressure* and hydrostatic pressure. The following discussion is intended to clarify these roles.

Each component gas in air exerts a measurable pressure, and the sum of these pressures constitutes atmospheric or barometric pressure, which is equivalent per unit of surface area at standard conditions to a pressure exerted by a column of mercury 760mm high or a column of water about 10 meters high (at sea level, excluding water vapor pressure). The pressure of an individual gas in air is called a *partial pressure*, and in water it is called a *tension*; both terms are an acknowledgement that the pressure of an individual gas is only part of the total atmospheric pressure. Likewise, each component gas will dissolve in water independently of all other gases, and when at equilibrium with the air, the pressure (tension) of a specific dissolved gas is equivalent to its partial pressure in the air. This relationship is evident in Table III–5 which lists the main constituents of dry air and their approximate partial pressures at sea level.

When supersaturation occurs, the diffusion pressure imbalance between the dissolved gas phase and the atmospheric phase favors a net transfer of gases from the water to the air. Generally this transfer cannot be accomplished fast enough by diffusion alone to prevent the formation of gas bubbles. However, a gas bubble cannot form in the water unless gas nuclei are present (Evans and Walder 1969,[116] Harvey et al. 1944b[122]) and unless the total dissolved gas pressure exceeds the sum of the compensating pressures such as hydrostatic pressure. Additional compensating pressures include blood pressure and viscosity, and their benefits may be significant.

Gas nuclei are probably unavoidable in surface water or in animals, because such nuclei are generated by any factor which decreases gas solubility, and because extreme measures are required to dissolve gas nuclei (Evans and Walder 1969;[116] Harvey et al. 1944b).[122] Therefore, hydrostatic pressure is a major preventive factor in gas bubble disease.

The effect of hydrostatic pressure is to oppose gas bubble formation. For example, one cannot blow a bubble out of a tube immersed in water until the gas pressure in the tube slightly exceeds the hydrostatic pressure at the end of the tube. Likewise a bubble cannot form in water, blood, or

TABLE III–5—Composition of Dry Air and Partial Pressures of Selected Gases at Sea Level

Gas	Molecular[a] percentage in dry air	Times atmospheric pressure	Individual gas[b] pressure in air or water at sea level
N_2...........	78.084	×760 mm Hg	=593.438 mm Hg
O_2...........	20.946	"	159.189 "
Ar...........	0.934	"	7.098 "
CO_2...........	0.033	"	0.250 "
Ne.....	0.00181	"	0.0138 "
He...........	0.00052	"	0.0039 "
			759.9927 mm Hg

[a] Glueckauf (1951[118]).
[b] At standard conditions excluding corrections for water vapor pressure.

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* In this Section gas tension will be called gas pressure and total gas tension will be called total dissolved gas pressure (TDGP). This is being done as a descriptive aid to readers who are not familiar with the terminology and yet need to convey these principles to laymen.

tissue until the total gas pressure therein exceeds the sum of atmospheric pressure (760mm Hg) plus hydrostatic pressure plus any other restraining forces. This relationship is illustrated in Figure III-1 which shows, for example, that gas bubbles could form in fresh water to a depth of about one meter when the total dissolved gas pressure is equal to 1.10 atmospheres; but they could not form below that point.

Excessive total dissolved gas pressure relative to ambient atmospheric pressure, therefore, represents a greater threat to aquatic organisms in the shallow but importantly productive littoral zone than in the deeper sublittoral zone. For example, if fish or their food organisms remain within a meter of the surface in water having a total dissolved gas pressure of 1.10 atmospheres, they are theoretically capable of developing gas bubble disease, especially if their body processes further decrease gas solubility by such means as physical activity, metabolic heat, increased osmolarity, or decreased blood pressure.

Hydrostatic pressure only opposes bubble formation; it does not decrease the kinetic energy of dissolved gas molecules except at extreme pressures. If this were not the case, aerobic animal life would be eliminated at or below a water depth equivalent to the pressure of oxygen, because there would be no oxygen pressure to drive O_2 across the gill membrane and thence into the blood. For a more detailed discussion of this subject, the reader is referred to Van Liere and Stickney's (1963)[138] and Randall's (1970a)[131] excellent reviews.

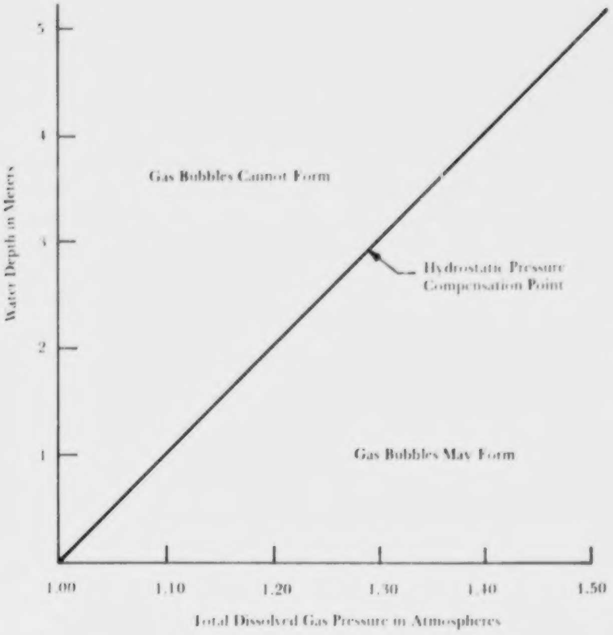

FIGURE III-1—Relationship of Total Dissolved Gas Pressure to Hydrostatic Pressure in Preventing Gas Bubble Formation

A final example will clarify the importance of total dissolved gas pressure. Eutrophic lakes often become supersaturated with photosynthetic dissolved oxygen, and such lakes commonly approach (or exceed) 120 per cent of saturation values for oxygen. But this only represents an additional dissolved gas pressure of about 32mm Hg ($O_2 = 159.19$ mm Hg$\times 0.2 = 31.83$ mm Hg) which equals:

$$\frac{760 \text{ mm Hg} + 31.83 \text{ mm Hg}}{760 \text{ mm Hg}} = 1.041 \text{ atmospheres of total dissolved gas pressure}$$

This imbalance apparently can be compensated in part by metabolic oxygen consumption, blood pressure, or both. On the other hand, a 1,000-fold increase in the neon saturation level would only increase the total dissolved gas pressure by about 1.8 mm Hg or:

$$\frac{1.8 \text{ mm Hg} + 760 \text{ mm Hg}}{760 \text{ mm Hg}} = 1.002 \text{ atmospheres}$$

This would not cause gas bubble disease.

The opposite situation can occur in spring water, where dissolved oxygen pressure is low and dissolved nitrogen and other gas pressures are high. In an actual case (Schneider *personal communication*),[144] dissolved nitrogen was reported to be 124 per cent of its air saturation value, whereas oxygen was 46 per cent of its air saturation value; total gas pressure was 1.046 of dry atmospheric pressure. Fish were living in this water, and although they probably suffered from hypoxia, they showed no symptoms of gas bubble disease.

How dissolved gases come out of solution and form bubbles (cavitate) is a basic physical and physiological topic which is only summarized here. Harvey et al. (1944b)[122] determined that bubble formation is promoted by boundary zone or surface interfaces which reduce surface tension and thereby decrease the dissolved gas pressure required for cavitation. For this reason, one usually sees gas bubbles forming first and growing fastest on submersed interfaces, such as tank walls, sticks, or the external surfaces of aquatic life.

Gas nuclei are apparently required for bubble formation, and these are considered to be ultra micro bubbles (Evans and Walder 1969).[116] These nuclei apparently represent an equilibrium between the extremely high compressive energy of surface tension and the pressure of contained gases. Lack of gas nuclei probably accounts for instances when extremely high but uncompensated dissolved gas pressures failed to cause bubble formation (Pease and Blinks 1947,[130] Hemmingsen 1970).[124] Gas nuclei are produced by anything that decreases gas solubility or surface tension (Harvey et al. 1944b,[122] Hills 1967,[125] Evans and Walder 1969)[116] and they can be eliminated at least temporarily by extremely high pressure which drives them back into solution (Evans and Walder 1969).[116]

Possible causes of gas nuclei formation in organisms include negative pressures in skeletal or cardiac muscle during pronounced activity (Whitaker et al. 1945),[141] eddy currents in the blood vascular system, synthetic or biologically produced surface-active compounds, and possible salting-out effects during hemoconcentration (as in saltwater adaptation). Once a bubble has formed, it grows via the diffusion of all gases into it.

Many factors influence the incidence and severity of gas bubble disease. For example, the fat content of an animal may influence its susceptibility. This has not been studied in fish, but Boycott and Damant (1908),[106] Behnke (1942),[102] and Gersh et al. (1944)[117] report that fat mammals are more susceptible than lean mammals to the "bends" in high-altitude decompression. This may be particularly significant to non-feeding adult Pacific salmon which begin their spawning run with considerable stored fat. This may also account in part for differences in the tolerances of different age groups or fish species. Susceptibility to gas bubble disease is unpredictable among wild fish, particularly when they are free to change their water depth and level of activity.

Gas Bubble Disease Syndrome and Effects

Although the literature documents many occurrences of gas bubble disease, data are usually missing for several important physical factors, such as hydrostatic pressure, barometric pressure, relative humidity, salinity, temperature, or other factors leading to calculation of total dissolved gas pressure. The most frequently reported parameter has been the calculated dissolved nitrogen (N_2) concentration or its percentage saturation from which one can estimate the pressure of inert gases. Thus the reported N_2 values provide only a general indication of the total dissolved gas pressure, which unfortunately tends to convey the erroneous concept that N_2 is the instigative or only significant factor in gas bubble disease.

Gas bubbles probably form first on the external surfaces of aquatic life, where total hydrostatic pressure is least and where an interface exists. Bubbles within the body of animals probably form later at low dissolved gas pressures, because blood pressure and other factors may provide additional resistance to bubble formation. However, at high dissolved gas pressure (>1.25 atm) bubbles in the blood may be the first recognizable symptom (Schneider *personal communication*).[144] In the case of larval fishes, zooplankton, or other small forms of aquatic life, the effect of external bubbles may be a blockage of the flow of water across the gills and asphyxiation or a change in buoyancy (Shirahata 1966).[136] The latter probably causes additional energy expenditure or floatation, causing potentially lethal exposure to ultraviolet radiation or potential predation.

The direct internal effects of gas bubble disease include a variety of symptoms that appear to be related primarily to the level of total dissolved gas pressure, the exposure time, and the *in vivo* location of lowest compensatory pressure.

The following is a résumé of Shirahata's (1966)[136] results. As the uncompensated total dissolved gas pressure increases, bubbles begin to appear on the fish, then within the skin, the roof of the mouth, within the fins, or within the abdominal cavity. Gas pockets may also form behind the eyeball and cause an exophthalmic "pop-eyed" condition. Probably gas emboli in the blood are the last primary symptoms to develop, because blood pressure and plasma viscosity oppose bubble formation. At some as yet undefined point, gas emboli become sufficiently large and frequent to cause hemostasis in blood vessels, which in turn may cause extensive tissue damage or complete hemostasis by filling the heart chamber with gas. The latter is the usual direct cause of death.

Exophthalmus or "pop-eye" and eye damage can be caused by several factors other than gas bubble disease and one should be duly cautious when tempted to diagnose gas bubble disease based solely on these criteria. While the above symptoms can be caused by excessive dissolved gas pressure (Westgard 1964),[140] they can also be caused by malnutrition, abrasion, and possibly by infection. Unfortunately there is no known definitive way to distinguish between latent eye damage caused by previous exposure to excessive dissolved gas pressure and other causes.

Secondary, latent, or sublethal effects of gas bubble disease in fish include promoting other diseases, necrosis, or other tissue changes, hemorrhages, blindness, and reproductive failure (Harvey and Cooper 1962,[123] Westgard 1964,[140] Pauley and Nakatani 1967,[129] and Bouck et al. 1971).[105] There is no known evidence that supersaturation causes a nitrogen narcosis in fish (such as can be experienced by scuba divers), as this requires high dissolved gas pressures probably above 10 atm. However, one can expect that fish afflicted with gas bubble disease or the above secondary effects might have their normal behavior altered.

There is no definitive evidence that fishes can detect supersaturation (Shelford and Allee 1913),[135] or that they actively avoid it by seeking hydrostatic pressure compensation (Ebel 1969).[112] However, the potential capacity to avoid supersaturation or to compensate by sounding is limited among anadromous species by the necessity of ascending their home river and by dams with relatively shallow fish ladders. This may also apply to other species that reproduce in or otherwise live in shallow-water niches. Physiological adaptation to supersaturation seems unlikely, and this contention is supported by the preliminary studies of Coutant and Genoway (1968).[109]

Interaction between gas bubble disease and other stresses is highly likely but not clearly established. Fish were more susceptible to a given level of total dissolved gas pressure when wounded (Egusa 1955).[114] The thermal tolerance of Pacific salmon was reduced when N_2 levels were 125 to 180 per cent in the case of juveniles (Ebel et al. 1971),[113] and when N_2 levels were >118 per cent in the case of adults (Coutant and Genoway 1968).[109] Chemicals or other factors that influence body activity or cardiovascular activity may also influence blood pressure (Randall 1970b),[132] and this would be expected to influence the degree to which the dissolved gas pressure is in excess, and hence the tolerance to gas bubble disease.

Variation in biological response is a prominent aspect of gas bubble disease, which should not be surprising in view of the numerous influential factors. Some of this variation might be explained by physiological differences between life stages or species, degree of fatness, blood pressure, blood viscosity, metabolic heat, body size, muscular activity, and blood osmolarity. For example, susceptibility to gas bubble disease may be inversely related to blood (or hemolymph) pressure. There is wide variation in blood pressure between life stages, between fish species, and between invertebrate species. Based on aortic blood pressures alone, one can hypothesize that largemouth bass (*Micropterus salmoides*) might be more susceptible to gas bubble disease than chinook salmon (*Oncorhynchus tshawytscha*) if other factors are equal. This contention is also supported by the observations that gas bubbles form in the blood of bullfrogs more easily than in rats (Berg et al. 1945),[104] possibly because of differences in blood pressure (Brand et al. 1951).[107]

Tolerance to supersaturation also varies between body sizes or life stages; Shirahata (1966)[136] relates this, in part, to an increase in cardiac and skeletal muscle activity. Larger fish were generally more sensitive to supersaturation than were smaller fish in most studies (Wiebe and McGavock 1932,[142] Egusa 1955,[114] Shirahata 1966,[136] Harvey and Cooper 1962).[123] Wood (1968)[143] has the opposite view, but he provides no supporting evidence. Possibly larger fish are more susceptible to gas bubble disease in part because they can develop greater metabolic heat than smaller fish. In this regard, Carey and Teal (1969)[108] reported that large tuna may have a muscle temperature as much as 10 C above the water temperature.

Data are quite limited on the tolerance of zooplankters and other aquatic invertebrates to excessive dissolved gas pressure. Evans and Walder (1969)[116] demonstrated that invertebrates can develop gas bubble disease. Unpublished observations by Nebeker* demonstrate that *Daphnia* sp. and *Gammarus* sp. are susceptible to gas bubble disease. On the other hand, it is widely known that some aquatic invertebrates are capable of diel migrations that may expose them to a considerable change in dissolved gas pressure; but apparently these organisms can tolerate or otherwise handle such changes. In view of the paucity of data, nothing firm can be said regarding the general tolerance of invertebrates to supersaturation.

Analytical Considerations

The apparatus and method of Van Slyke et al. (1934)[139] are still the standard analytical tools for most gas analyses. Scholander et al. (1955)[134] and others have developed similar methods with modifications to accomodate their special needs. More recently, Swinnerton et al. (1962)[137] published a gas analysis method that utilizes gas-liquid chromatography. However, both of these basic methods have drawbacks, because they either require special expertise or do not otherwise meet the field needs of limnologists and fisheries or pollution biologists.

A new device by Weiss* measures the differential gas pressure between the air and the water within fifteen minutes. This portable device is simple to operate, easy and inexpensive to build, and gives direct readings in mm Hg. Unpublished data by Weiss show that this instrument has an accuracy comparable to the Van Slyke and the chromatographic procedures. The instrument consists of a gas sensor (150 ft. coil of small diameter, thin-walled, silicone rubber tube) connected to a mercury manometer. The sensor is placed underwater where the air in the tubing equilibrates with the dissolved gases in the water. The resulting gas pressure is read directly via the mercury manometer which gives a positive value for supersaturated water and a negative value for water that is not fully saturated.

Total Dissolved Gas Pressure Criteria

Safe upper limits for dissolved gases must be based on the total dissolved gas pressures (sum of all gas tensions) and not solely on the saturation value of dissolved nitrogen gas alone. Furthermore, such limits must provide for the safety of aquatic organisms that inhabit or frequent the shallow littoral zone, where an existing supersaturation could be worsened by heating, photosynthetic oxygen production, or other factors. There is little information on the chronic sublethal effects of gas bubble disease and almost all the research has been limited to species of the family Salmonidae. Likewise, gas tolerance data are unavailable for zooplankters and most other aquatic invertebrates. Therefore, it is necessary to judge safe limits from data on mortality of selected salmonid fishes that were held under conditions approximating the shallow water of a hypothetical littoral zone. These data are:

1. Shirahata (1966)[136] reported that advanced fry of rainbow trout (*Salmo gairdneri*) experienced 10 per cent mortality when N_2 was about 111 per cent of its saturation value. He concludes that, "... the nitrogen contents which did not cause any gas disease were ... less than 110 per cent to the more advanced fry."

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* Dr. Ray Weiss, University of California, Scripps, Institute of Oceanography, Geological Research Division, P. O. Box 109, La Jolla, California 92037.

2. Harvey and Cooper (1962)[123] reported that fry of sockeye salmon (*Oncorhynchus nerka*) suffered latent effects (necrosis and hemorrhages) for some time after normal gas levels were said to have been restored.

3. Coutant and Genoway (1968)[109] reported that sexually precocious spring chinook salmon (*Oncorhynchus tshawytscha*) weighing 2 to 4 kg, experienced extensive mortality in six days when exposed at or above 118 per cent of N_2 saturation; these salmon experienced no mortality when N_2 was below 110 per cent of saturation.

Whether or not other species or life stages of aquatic life may be more or less sensitive than the above salmonids remains to be proven. In the meantime, the above references provide the main basis for establishing the following total dissolved gas recommendations.

Recommendations

Available data for salmonid fish suggest that aquatic life will be protected only when *total* dissolved gas pressure in water is no greater than 110 per cent of the existing atmospheric pressure. Any prolonged artificial increase in total dissolved gas pressure should be avoided in view of the incomplete body of information.

CARBON DIOXIDE

Carbon dioxide exists in two major forms in water. It may enter into the bicarbonate buffering system at various concentrations depending on the pH of the water. In addition, "free" carbon dioxide may also exist, and this component affects the respiration of fish (Fry 1957).[151] Because of respiratory effects, free carbon dioxide is the form considered most significant to aquatic life.

The concentration of free carbon dioxide, where oxygen-demanding wastes are not excessive, is a function of pH, temperature, alkalinity, and the atmospheric pressure of carbon dioxide. Doudoroff (1957)[147] reported that concentrations of free carbon dioxide above 20 mg/l occur rarely, even in polluted waters; and Ellis (1937)[150] found that the free carbon dioxide content of Atlantic Coast streams ranged between zero and 12 mg/l. Ellis (1937)[150] and Hart (1944)[152] both reported that in 90 to 95 per cent of the fresh waters in the United States that support a good and diverse fish population the free carbon dioxide concentrations fall below 5 mg/l.

An excess of free carbon dioxide may have adverse effects on aquatic life. Powers and Clark (1943)[156] and Warren (1971)[157] reported that fish are able to detect and to respond to slight gradients in carbon dioxide tension. Brinley (1943)[146] and Höglund (1961)[154] observed that fish may avoid free carbon dioxide levels as low as 1.0 to 6.0 mg/l.

Elevated carbon dioxide concentrations may interfere with the ability of fish to respire properly and may thus affect dissolved oxygen uptake. Doudoroff and Katz (1950)[148] and Doudoroff and Shumway (1970)[149] reported that where dissolved oxygen uptake interference does occur, the free carbon dioxide concentrations which appreciably affect this are higher than those found in polluted waters. In bioassay tests using ten species of warmwater fish, Hart (1944)[152] found that the gizzard shad (*Dorosoma cepedianum*) was the most sensitive and was unable to remove oxygen from water 50 per cent saturated with dissolved oxygen in the presence of 88 mg/l of free carbon dioxide. The less sensitive, largemouth bass (*Micropterus salmoides*) was unable to extract oxygen when the carbon dioxide level reached 175 mg/l. Below 60 mg/l of free carbon dioxide, most species of fish had little trouble in extracting dissolved oxygen from the water.

High concentrations of free carbon dioxide cause pronounced increases in the minimum dissolved oxygen requirement of coho salmon (*Oncorhynchus kisutch*), but these fish acclimatized rapidly to carbon dioxide concentrations as high as 175 mg/l at 20 C when the dissolved oxygen level was near saturation (McNeil 1956).[155]

Basu (1959)[145] found that for most fish species, carbon dioxide affected the fishes' ability to consume oxygen in a predictable manner. He further indicated that temperature affected carbon dioxide sensitivity, being less at higher water temperatures.

The ability of fish to acclimatize to increases in carbon dioxide concentrations as high as 60 mg/l with little effect has been indicated by Haskell and Davies (1958).[153] Doudoroff and Shumway (1970)[149] indicate that the ability of fish to detect low free carbon dioxide concentrations, the presence of low carbon dioxide levels in most waters, and the ability of fish to acclimatize to carbon dioxide in the water probably prevent this constituent from becoming a major hazard.

Recommendation

Concentrations of free carbon dioxide above 20 mg/l occur rarely. Fish acclimatize to increases in carbon dioxide levels as high as 60 mg/l with little effect. However, fish are able to detect and respond to slight gradients and many avoid free carbon dioxide levels as low as 1.0 to 6.0 mg/l.

ACIDITY, ALKALINITY, AND pH

NATURAL CONDITIONS AND SIGNIFICANCE

Acidity in natural waters is caused by carbon dioxide, mineral acids, weakly dissociated acids, and the salts of strong acids and weak bases. The alkalinity of a water is actually a measure of the capacity of the carbonate-bicarbonate system to buffer the water against change in pH. Technical information on alkalinity has recently been reviewed by Kemp (1971).[162]

An index of the hydrogen ion activity is pH. Even though pH determinations are used as an indication of acidity or alkalinity or both, pH is not a measure of either. There is a relationship between pH, acidity, and alkalinity (Standard Methods 1971):[164] water with a pH of 4.5 or lower has no measurable alkalinity, and water with a pH of 8.3 or higher has no measurable acidity. In natural water, where the pH may often be in the vicinity of 8.3, acidity is not a factor of concern. In most productive fresh waters, the pH falls in a range between 6.5 and 8.5 (except when increased by photosynthetic activity). Some regions have soft waters with poor buffering capacity and naturally low pH. They tend to be less productive. Such conditions are found especially in dark colored waters draining from coniferous forests or muskegs, and in swampy sections of the Southeast. For a variety of reasons, some waters may exhibit quite extreme pH values. Before these are considered natural conditions, it should be ascertained that they have not actually resulted from man-made changes, such as stripping of ground cover or old mining activities. This is important because the recommendations refer to estimated natural levels.

TOXICITY TO AQUATIC LIFE

Some aquatic organisms, especially algae, have been found to live at pH 2 and lower, and others at pH 10 and higher; however, such organisms are relatively few. Some natural waters with a pH of 4 support healthy populations of fish and other organisms. In these cases the acidity is due primarily to carbon dioxide and natural organic acids, and the water has little buffering capacity. Other natural waters with a pH of 9.5 also support fish but are not usually highly productive.

The effects of pH on aquatic life have been reviewed in detail in excellent reports by the European Inland Fisheries Advisory Commission (1969)[160] and Katz (1969).[161] Interpretations and summaries of these reviews are given in Table III-6.

ADVERSE INDIRECT EFFECTS OR SIDE EFFECTS

Addition of either acids or alkalies to water may be harmful not only by producing acid or alkaline conditions, but also by increasing the toxicity of various components in the waters. For example, acidification of water may release free carbon dioxide. This exerts a toxic action additional to that of the lower pH. Recommendations for pH are valid if carbon dioxide is less than 25 mg/l (see the discussion of Carbon Dioxide, p. 139).

A reduction of about 1.5 pH units can cause a thousand-fold increase in the acute toxicity of a metallocyanide complex (Doudoroff et al. 1966).[159] The addition of strong alkalies may cause the formation of undissociated NH_4OH or un-ionized NH_3 in quantities that may be toxic (Lloyd 1961,[163] Burrows 1964).[155] Many other pollutants may change their toxicity to a lesser extent. It is difficult to predict whether toxicity will increase or decrease for a given direction of change in pH.

Weakly dissociated acids and bases must be considered in terms of their toxicities, as well as their effects on pH and alkalinity.

The availability of many nutrient substances varies with the hydrogen ion concentration. Some trace metals become more soluble at low pH. At higher pH values, iron tends to become unavailable to some plants, and hence the production of the whole aquatic community may be affected.

The major buffering system in natural waters is the carbonate system that not only neutralizes acids and bases to reduce the fluctuations in pH, but also forms a reservoir of carbon for photosynthesis. This process is indispensable, because there is a limit on the rate at which carbon dioxide can be obtained from the atmosphere to replace that in the water. Thus the productivity of waters is closely correlated to the carbonate buffering system. The addition of mineral acids preempts the carbonate buffering capacity, and the

TABLE III–6—A Summary of Some Effects of pH on Freshwater Fish and Other Aquatic Organisms

pH	Known effects
11.5-12.0	Some caddis flies (Trichoptera) survive but emergence reduced.
11.0-11.5	Rapidly lethal to all species of fish.
10.5-11.0	Rapidly lethal to salmonids. The upper limit is lethal to carp (Cyprinus carpio), goldfish (Carassius auratus), and pike. Lethal to some stoneflies (Plecoptera) and dragonflies (Odonata). Caddis fly emergence reduced.
10.0-10.5	Withstood by salmonids for short periods but eventually lethal. Exceeds tolerance of bluegills (Lepomis macrochirus) and probably goldfish. Some typical stoneflies and mayflies (Ephemera) survive with reduced emergence.
9.5-10.0	Lethal to salmonids over a prolonged period of time and no viable fishery for coldwater species. Reduces populations of warmwater fish and may be harmful to development stages. Causes reduced emergence of some stoneflies.
9.0-9.5	Likely to be harmful to salmonids and perch (Perca) if present for a considerable length of time and no viable fishery for coldwater species. Reduced populations of warmwater fish. Carp avoid these levels.
8.5-9.0	Approaches tolerance limit of some salmonids, whitefish (Coregonus), catfish (Ictaluridae), and perch. Avoided by goldfish. No apparent effects on invertebrates.
8.0-8.5	Motility of carp sperm reduced. Partial mortality of burbot (Lota lota) eggs.
7.0-8.0	Full fish production. No known harmful effects on adult or immature fish, but 7.0 is near low limit for Gammarus reproduction and perhaps for some other crustaceans.
6.5-7.0	Not lethal to fish unless heavy metals or cyanides that are more toxic at low pH are present. Generally full fish production, but for fathead minnow (Pimephales promelas), frequency of spawning and number of eggs are somewhat reduced. Invertebrates except crustaceans relatively normal, including common occurrence of mollusks. Microorganisms, algae, and higher plants essentially normal.
6.0-6.5	Unlikely to be toxic to fish unless free carbon dioxide is present in excess of 100 ppm. Good aquatic populations with varied species can exist with some exceptions. Reproduction of Gammarus and Daphnia prevented, perhaps other crustaceans. Aquatic plants and microorganisms relatively normal except fungi frequent.
5.5-6.0	Eastern brook trout (Salvelinus fontinalis) survive at over pH 5.5. Rainbow trout (Salmo gairdneri) do not occur. In natural situations, small populations of relatively few species of fish can be found. Growth rate of carp reduced. Spawning of fathead minnow significantly reduced. Mollusks rare.
5.0-5.5	Very restricted fish populations but not lethal to any fish species unless CO_2 is high (over 25 ppm), or water contains iron salts. May be lethal to eggs and larvae of sensitive fish species. Prevents spawning of fathead minnow. Benthic invertebrates moderately diverse, with certain black flies (Simulidae), mayflies (Ephemerella), stoneflies, and midges (Chironomidae) present in numbers. Lethal to other invertebrates such as the mayfly. Bacterial species diversity decreased; yeasts and sulfur and iron bacteria (Thiobacillus-Ferrobacillus) common. Algae reasonably diverse and higher plants will grow.
4.5-5.0	No viable fishery can be maintained. Likely to be lethal to eggs and fry of salmonids. A salmonid population could not reproduce. Harmful, but not necessarily lethal to carp. Adult brown trout (Salmo trutta) can survive in peat waters. Benthic fauna restricted, mayflies reduced. Lethal to several typical stoneflies. Inhibits emergence of certain caddis fly, stonefly, and midge larvae. Diatoms are dominant algae.
4.0-4.5	Fish populations limited; only a few species survive. Perch, some coarse fish, and pike can acclimate to this pH, but only pike reproduce. Lethal to fathead minnow. Some caddis flies and dragonflies found in such habitats; certain midges dominant. Flora restricted.
3.5-4.0	Lethal to salmonids and bluegills. Limit of tolerance of pumkinseed (Lepomis gibbosus), perch, pike, and some coarse fish. All flora and fauna severely restricted in number of species. Cattail (Typha) is only common higher plant.
3.0-3.5	Unlikely that any fish can survive for more than a few hours. A few kinds of invertebrates such as certain midges and alderflies, and a few species of algae may be found at this pH range and lower

original biological productivity is reduced in proportion to the degree that such capacity is exhausted. Therefore, the minimum essential buffering capacity and tolerable pH limits are important water quality considerations.

Because of this importance, there should be no serious depletion of the carbonate buffering capacity, and it is recommended that reduction of alkalinity of natural waters should not exceed 25 per cent.

Recommendations

Suggested maximum and minimum levels of protection for aquatic life are given in the following recommendations. A single range of values could not apply to all kinds of fish, nor could it cover the different degrees of graded effects. The selection of the level of protection is a socioeconomic decision, not a biological one. The levels are defined in Table III–3 (see the discussion of Dissolved Oxygen).

Nearly Maximum Level of Protection

- **pH not less than 6.5 nor more than 8.5. No change greater than 0.5 units above the estimated natural seasonal maximum, nor below the estimated natural seasonal minimum.**

High Level of Protection

- **pH not less than 6.0 nor more than 9.0. No change greater than 0.5 units outside the estimated natural seasonal maximum and minimum.**

Moderate Level of Protection

- **pH not less than 6.0 nor more than 9.0. No change greater than 1.0 units outside the estimated natural seasonal maximum and minimum.**

Low Level of Protection

- **pH not less than 5.5 nor more than 9.5. No change greater than 1.5 units outside the estimated natural seasonal maximum and minimum.**

Additional Requirements for All Levels of Protection

- **If a natural pH is outside the stated range of pH for a given level of protection, no further change is desirable.**
- **The extreme range of pH fluctuation in any location should not be greater than 2.0 units. If natural fluctuation exceeds this, pH should not be altered.**
- **The natural daily and seasonal patterns of pH variation should be maintained, although the absolute values may be altered within the limits specified.**
- **The total alkalinity of water is not to be decreased more than 25 per cent below the natural level.**

DISSOLVED SOLIDS AND HARDNESS

Surface water at some time and place may contain a trace or more of any water-soluble substance. The significance and the effects of small concentrations of these substances are discussed separately throughout this Report. The presence and relative abundance of these constituents in water is influenced by several factors, including surface runoff, geochemistry of the watershed, atmospheric fallout including snow and rainfall, man-created effluents, and biological and chemical processes in the water itself. Many of these dissolved materials are essential to the life processes of aquatic organisms. For a general discussion of the chemistry of fresh water the reader is referred to Hutchinson (1957)[167] and Ruttner (1963).[172]

A general term describing the concentration of dissolved materials in water is *total dissolved solids.* The more conspicuous constituents of total dissolved solids in natural surface waters include carbonates, sulfates, chlorides, phosphates, and nitrates. These anions occur in combination with such metallic cations as calcium, sodium, potassium, magnesium, and iron to form ionizable salts (Reid 1961).[170]

Concentrations and relative proportions of dissolved materials vary widely with locality and time. Hart et al. (1945)[166] reported that in the inland waters of the United States which support a mixed biota, 5 per cent have a dissolved solids concentration under 72 mg/l; about 50 per cent under 169 mg/l; and 95 per cent under 400 mg/l. Table III–7 provides information on ranges and median concentrations of the major ions in United States streams.

The quantity and quality of dissolved solids are major factors in determining the variety and abundance of plants and animals in an aquatic system. They serve as nutrients in productivity, osmotic stress, and direct toxicity. A major change in quantity or composition of total dissolved solids changes the structure and function of aquatic ecosystems. Such changes are difficult to predict.

Concentrations of dissolved solids affecting freshwater fish by osmotic stress are not well known. Mace (1953)[169] and Rounsefell and Everhart (1953)[171] reported that the upper limit may range between 5,000 and 10,000 mg/l total dissolved solids, depending on species and prior acclimatization. The literature indicates that concentrations of total dissolved solids that cause osmotic stress in adult fish are higher than the concentrations existing in most fresh waters of the United States. Many dissolved materials are toxic at concentrations lower than those where osmotic effect can be expected. (See Toxic Substances, p. 172, and Acidity, Alkalinity, and pH, p. 140.)

Hardness of surface waters is a component of total dissolved solids and is chiefly attributable to calcium and magnesium ions. Other ions such as strontium, barium, manganese, iron, copper, zinc, and lead add to hardness, but since they are normally present in minor concentrations their effect is usually minimal. Generally, the biological productivity of a water is directly correlated with its hardness. However, while calcium and magnesium contribute to hardness and productivity, many other elements (when present in concentrations which contribute a substantial measure of hardness) reduce biological productivity and are toxic. Hardness per se has no biological significance because biological effects are a function of the specific concentrations and combinations of the elements present.

The term "hardness" serves a useful purpose as a general index of water type, buffering capacity, and productivity. Waters high in calcium and magnesium ions (hard water) lower the toxicity of many metals to aquatic life (Brown 1968;[165] Lloyd and Herbert 1962).[168] (See Figure III–9 in the discussion of Metals, p. 178.) However, the term "hardness" should be avoided in delineating water quality

TABLE III–7—Major Dissolved Constituents of River Waters Representing About 90 Percent of Total Stream Flow in the United States

Constituent	Median mg/l	Range mg/l
Total dissolved solids	169	72–400
Bicarbonate (HCO_3)	90	40–180
Sulfate (SO_4)	32	11–90
Chloride (Cl)	9	3–170
Calcium (Ca)	28	15–52
Magnesium (Mg)	7	3.5–14
Sodium and potassium (Na and K)	10	6–85

Source: After Hart et al. (1945)[166]

requirements for aquatic life. More emphasis should be placed on specific ions.

Recommendation

Total dissolved materials should not be changed to the extent that the biological communities characteristic of particular habitats are significantly changed. When dissolved materials are altered, bioassays and field studies can determine the limits that may be tolerated without endangering the structure and function of the aquatic ecosystem.

OILS

Losses of oil that can have an adverse effect on water quality and aquatic life can occur in many of the phases of oil production, refining, transportation, and use. Pollution may be in the form of floating oils, emulsified oils, or solution of the water soluble fraction of these oils.

The toxicity of crude oil has been difficult to interpret since crude oil may contain many different organic compounds and inorganic elements. The composition of such oils may vary from region to region, and petroleum products produced can be drastically different in character in line with their different intended uses (Purdy 1958).[198] The major components of crude oil can be categorized as aliphatic normal hydrocarbons, cyclic paraffin hydrocarbons, aromatic hydrocarbons, naphtheno-aromatic hydrocarbons, resins, asphaltenes, heteroatomic compounds, and metallic compounds (Bestougeff 1967).[175] The aromatic hydrocarbons in crude oil appear to be the major group of acutely toxic compounds (Blumer 1971,[176] Shelton 1971).[199]

Because the biological effects of oils and the relative merits of control measures are discussed in detail in Section IV (p. 257) of this Report, only effects of special interest or pertinence to fresh water are discussed here. The effects of floating oil on wildlife are discussed on p. 196.

OIL REFINERY EFFLUENTS

Copeland and Dorris (1964)[180] studied primary productivity and community respiration in a series of oil refinery effluent oxidation ponds. These ponds received waste waters which had been in contact with the crude oil and various products produced within the refinery. Surface oils were skimmed. In the series of oxidation ponds, primary productivity and community respiration measurements clearly indicated that primary producers were limited in the first ponds, probably by toxins in the water. Oxidation ponds further along in the series typically supported algal blooms. Apparently degradation of the toxic organic compounds reduced their concentration below the threshold lethal to the algae. Primary productivity was not greater than community respiration in the first ponds in the series. Minter (1964)[195] found that species diversity of phytoplankton was lowest in the first four ponds of the series of ten. A "slug" of unknown toxic substance drastically reduced the species diversity in all ponds. Zooplankton volumes increased in the latter half of the pond series, presumably as a result of decreasing toxicity. Benthic fauna species diversity in streams receiving oil refinery effluents was low near the outfall and progressively increased downstream as biological assimilation reduced the concentration of toxins (Wilhm and Dorris 1966,[206] Harrel et al. 1967,[184] Mathis and Dorris 1968[191]).

Long-term, continuous-flow bioassays of biologically treated oil refinery effluents indicated that complex refineries produce effluents which contain cumulative toxins of substances that cause accumulative deleterious effects (Graham and Dorris 1968).[182] Subsequent long-term continuous-flow bioassays of biologically treated oil refinery effluents indicated that passage of the effluent through activated carbon columns does not remove the fish toxicants. Of the fathead minnows (*Pimphales promelas*) tested, half were killed in 14 days, and only 10 per cent survived 30 days (Burks 1972[207] *personal communication*). Trace organic compounds identified in extracts from the effluent were a homologous series of aliphatic hydrocarbons ($C_{11}H_{22}$ through $C_{18}H_{38}$) and isomers of cresol and xylenol. Since the soluble fractions derived from oil refineries are quantitatively, and to some extent qualitatively, different from those derived from oil spills, care must be taken to differentiate between these two sources.

FREE AND FLOATING OIL

Free oil or emulsions may adhere to the gills of fish, interfering with respiration and causing asphyxia. Within limits, fish are able to combat this by defensive mucous secretions (Cole 1941).[179] Free oil and emulsions may likewise coat aquatic plants and destroy them (McKee and Wolf 1963).[193]

Fish and benthic organisms may be affected by soluble substances extracted from the oils or by coating from emulsified oils. Water soluble compounds from crude or manufactured oils may also contain tainting substances which affect the taste of fish and waterfowl (Krishnawami and Kupchanko 1969).[189]

Toxicity tests for oily substances provide a broad range of results which do not permit rigorous safety evaluations. The variabilities are due to differences in petroleum products tested, non-uniform testing procedures, and species differences. Most of the research on the effects of oils on aquatic life has used pure compounds which exist only in low percentages in many petroleum products or crude oils. Table III–8 illustrates the range of reported toxicities. For halo-, nitro-, or thio-derivatives, the expected toxicity would be greater.

Because of the basic difficulties in evaluating the toxicity, especially of the emulsified oils, and because there is some evidence that oils may persist and have subtle chronic effects (Blumer 1971),[176] the maximum allowable concentration of emulsified oils should be determined on an individual basis and kept below 0.05 of the 96-hour LC50 for sensitive species.

TABLE III–8—Toxicity Ranges

Chemical	ppm. conc.	Effect	Species	Investigator
Aniline	379	none	Daphnia magna	Anderson 1944[17]
Benzene	31	96 hr LC50	Pimephales promelas	Pickering & Henderson 1966[197]
	22	96 hr LC50	Lepomis macrochirus	" "
	32	96 hr LC50	Carassius auratus	" "
Cresol	10	96 hr LC50	Lepomis macrochirus	Cairns & Scheier 1959[178]
Cyclohexane	30	96 hr LC50	Pimephales promelas	Pickering & Henderson 1966[197]
	31	96 hr LC50	Lepomis macrochirus	" "
	33	"	Carassius auratus	" "
	48	"	Lebistes reticulatus	" "
Ethylbenzene	40	"	Pimephales promelas	" "
	29	"	Lepomis macrochirus	" "
	73	"	Carassius auratus	" "
	78	"	Lebistes reticulatus	" "
Heptane	4924	48 hr LC50	Gambusia affinis	Wallen et al. 1957[205]
Isoprene	75	96 hr LC50	Pimephales promelas	Pickering & Henderson 1966[197]
	39	"	Lepomis macrochirus	" "
	180	"	Carassius auratus	" "
	140	"	Lebistes reticulatus	" "
Nephenic acid	5.6	"	Lepomis macrochirus	Cairns & Scheier 1958[177]
	6.6–7.5	"	Physa heterostropha	" "
Naphthalene	165	48 hr LC50	Gambusia affinis	Wallen et al. 1957[205]
Toluene	1260	"	"	" "
	44	96 hr LC50	Pimephales promelas	Pickering & Henderson 1966[197]
	24	"	Lepomis macrochirus	" "
	62	"	Carassius auratus	" "
	66	"	Lebistes reticulatus	" "
Gasoline	91	48 hr LC50	Alosa sapidissima	Tagatz 1961[203]
	40	96 hr LC50	Salmo gairdneri	Meinck et al. 1956[194]
Cutting oil #2	14,500	96 hr LC50	Salmo gairdneri	Turnbull et al 1954[204]
Diesel fuel	167	48 hr LC50	Alosa sapidissimia	Tagatz 1961[203]
Bunker oil	2417	"	Alosa sapidissima	" "
Bunker C oil	1700	168 hr LC50	Salmo salar	Sprague and Carson manuscript 1970[200]

SEDIMENTED OIL

Ludzack et al. (1957)[190] found that the sediment in the Ottawa River in Ohio downstream from a refinery consisted of up to 17.8 per cent oil. Hunt (1957)[187] and Hartung and Klingler (1968)[185] reported on the occurrence of sedimented oil in the Detroit River. North et al. (1965)[196] found sedimented oils after an oil pollution incident in marine coves in Baja California. Forbes and Richardson (1913)[181] reported 2.5 per cent oils in the bottom deposits of the Illinois River. McCauley (1964)[192] reported finding oily bottom deposits after oil pollution near Boston. Thus, while the reports may be scattered, the evidence is clear that the existence of sedimented oils in association with oil pollution is widespread.

There is an increasing body of evidence indicating that aliphatic hydrocarbons are synthesized by aquatic organisms and find their way into sediments in areas which have little or no history of oil pollution (Han et al. 1968,[183] Avigan and Blumer 1968[174]). Hydrocarbons have been reported in the recent sediments of lakes in Minnesota (Swain 1956)[202] and the Gulf of Mexico (Stevens et al. 1956).[201]

Areas which contain oily sediments usually have an impoverished benthic fauna; it is not clear to what extent oil contributes to this, because of the presence of other pollutants (Hunt 1962).[188] However, there are recurring reports of a probable relationship between sedimented oils and altered benthic communities. Sedimented oils may act as concentrators for chlorinated hydrocarbon pesticides (Hartung and Klingler 1970),[186] but the biological implications indicate that additional study is required.

Because of the differences in toxicities of sedimented oils and because of limited knowledge on quantities which are harmful to aquatic life, it is suggested that the concentration of hexane extractable substances (exclusive of elemental sulfur) in air-dried sediments not be permitted to increase above 1,000 mg/kg on a dry weight basis.

Recommendations

Aquatic life and wildlife should be protected where:

- **there is no visible oil on the surface;**
- **emulsified oils do not exceed 0.05 of the 96-hour LC50;**
- **concentration of hexane extractable substances (exclusive of elemental sulfur) in air-dried sediments does not increase above 1,000 mg/kg on a dry weight basis.**

TAINTING SUBSTANCES

Discharges from municipal wastewater treatment plants, a variety of industrial wastes and organic compounds, as well as biological organisms, can impart objectionable taste, odor, or color to the flesh of fish and other edible aquatic organisms. Such tainting can occur in waters with concentrations of the offending material lower than those recognized as being harmful to an animal (Tables III–9 and III–10).

BIOLOGICAL CAUSES OF TAINTING

Thaysen (1935)[231] and Thaysen and Pentelow (1936)[232] demonstrated that a muddy or earthy taste can be imparted to the flesh of trout by material produced by an odiferous species of *Actinomyces*. Lopinot (1962)[224] reported a serious fish and municipal water supply tainting problem on the Mississippi River in Illinois during a period when actinomycetes, *Oscillatoria*, *Scenedesmus*, and *Actinastrum* were abundant. *Oscillatoria princeps* and *O. agardhi* in plankton of a German lake were reported by Cornelius and Bandt (1933)[213] as causing off-flavor in lake fish. Aschner et al. (1967)[210] concluded that the benthic alga, *O. tenuis*, in rearing ponds in Israel was responsible for imparting such a bad flavor to carp (*Cyprinus carpio*) that the fish were unacceptable on the market. Henley's (1970)[221] investigation of odorous metabolites of Cyanophyta showed that *Anabaena circinalis* releases geosmin and indicated that this material was responsible for the musty or earthy odor often characteristic of water from reservoirs with heavy algal growths in summer and fall.

Oysters occasionally exhibit green coloration of the gills due to absorption of the blue-green pigment of the diatom, *Navicula*, (Ranson 1927).[225]

TAINTING CAUSED BY CHEMICALS

Phenolic compounds are often associated with both water and fish tainting problems (Table III–9). However, Albersmeyer (1957)[208] and Albersmeyer and Erichsen (1959)[209] found that, after being dephenolated, both a carbolated oil and a light oil still imparted a taste to fish more pronounced than that produced by similar exposures to naphthalene and methylnaphthalene (phenolated compounds). They concluded that other hydrocarbons in the oils were more responsible for imparting off-flavor than the phenolic materials in the two naphthalenes tested.

Refineries (Fetterolf 1962),[215] oily wastes (Zillich 1969),[236] and crude oil (Galtsoff et al. 1935)[219] have been associated with off-flavor problems of fish and shellfish in both freshwater and marine situations (Westman and Hoff 1963).[234] Krishnawami and Kupchanko (1969)[223] demonstrated that rainbow trout (*Salmo gairdneri*) adsorbed enough compounds from a stream polluted with oil slicks and oil refinery effluents to exhibit a definite oily taste and flavor. In waters receiving black liquor from kraft pulp mills, the gills and mantles of oysters developed a gray color (Galtsoff et al. 1947).[218] The authors also found this condition in oysters grown in waters receiving domestic sewage. Newton (1967)[237] confined trout in live-cages and correlated inten-

TABLE III–9—Wastewaters Found to have Lowered the Palatability of Fish Flesh

Wastewater source	Concentration in water affecting palatability of fish	Species	Reference
2,4-D mfg. plant	50–100 mg/l	Trout	Shumway 1966[228]
Coal—coking	0.02–0.1 mg/l	Freshwater fish	Bandt 1955[211]
Coal—tar	0.1 mg/l	Freshwater fish	Bandt 1955[211]
Kraft process (untreated)	1–2% by vol.	Salmon	Shumway and Chadwick 1971[229]
Kraft process (treated)	9–12% by vol.	Salmon	Shumway and Palensky, unpublished data[239]
Kraft and neutral sulfite process	....	Trout	Newton 1967[237]
Municipal dump runoff	....	Channel catfish (Ictalurus punctatus)	Thomas and Hicks 1971[233]
Municipal untreated sewage (2 locations)	....	Channel catfish	Thomas and Hicks 1971[233]
Municipal wastewater treatment plants (4 locations)	....	Channel catfish	Thomas and Hicks 1971[233]
Municipal wastewater treatment plant (Primary)	11–13% by vol.	Freshwater fish	Shumway and Palensky, unpublished data[239]
Municipal wastewater treatment plant (Secondary)	20–26% by vol.	Freshwater fish	Shumway and Palensky, unpublished data[239]
Oily wastes	....	Trout	Zillich 1969[236]
Refinery	....	Trout	Fetterolf 1962[216]
Sewage containing phenols	0.1 mg/l	Freshwater fish	Bandt 1955[211]
Slaughterhouses (2 locations)	....	Channel catfish	Thomas and Hicks 1971[233]

TABLE III-10—Concentrations of Chemical Compounds in Water That Can Cause Tainting of the Flesh of Fish and Other Aquatic Organisms

Chemical	Estimated threshold level in water (mg/l)	Reference*
acetophenone	0.5	d
acrylonitrile	18	g
cresol	0.07	g
m-cresol	0.2	g
o-cresol	0.4	g
p-cresol	0.12	g
cresylic acid (meta para)	0.2	d
N-butylmercaptan	0.06	g
p-sec. butylphenol	0.3	d
p-tert. butylphenol	0.03	d
o-chlorophenol	0.0001 to 0.015	b, d, e
p-chlorophenol	0.01 to 0.05	d, g, e
2,3-dichlorophenol	0.084	g
2,4-dichlorophenol	0.001 to 0.014	d, f, g
2,5-dichlorophenol	0.023	g
2,6-dichlorophenol	0.035	g
2-methyl, 4-chlorophenol	0.075	g
2-methyl, 6-chlorophenol	0.003	g
o-phenylphenol	1	d
2,4,6-trichlorophenol	0.003 to 0.05	g
phenol	1 to 10	d, e
phenols in polluted river	0.02 to 0.15	a
diphenyl oxide	0.05	d
β,β-dichlorodiethyl ether	0.09 to 1.0	d, g
o-dichlorobenzene	0.25	d
ethylbenzene	<0.25	d
ethanethiol	0.24	g
ethylacrylate	0.6	g
formaldehyde	95	g
kerosene	0.1	d
kerosene plus kaolin	1	i
isopropylbenzene	<0.25	d
naphtha	0.1	d
naphthalene	1	a
naphthol	0.5	a
2-naphthol	0.3	g
dimethylamine	7	g
α-methylstyrene	0.25	d
oil, emulsifiable	>15	d
pyridine	5 to 28	a, g
pyrocatechol	0.8 to 5	a, g
pyrogallol	20 to 30	a
quinoline	0.5 to 1	a
p-quinone	0.5	a
styrene	0.25	d
toluene	0.25	d
outboard motor fuel, as exhaust	2.6 gal/acre-foot	c, h
guaiacol	0.082	g

* Reference key:
a Bandt 1955[211]
b Boetius 1954[212]
c English et al. 1963[214]
d Fetterolf 1964[215] published the results of A. W. Winston, Jr. of the Dow Chemical Company. The data are also available in an undated mimeographed release of the company
e Schulze 1961[227]
f Shumway 1966[228]
g Shumway, D. L. and J. R. Palensky,[239] unpublished data (1971).
h Surber, et al. 1965[230]
i Westman and Hoff 1963[234]

sity of off-flavor with proximity to the discharge of a paper mill using both the neutral sulfite and kraft processes.

Shellfish have the ability to concentrate and store metals at levels greater than the concentrations in the water (see Section I, pp. 36–37, and Section IV, p. 240). Oyster flesh can become green-colored from copper accumulation. The copper content of normal-colored oyster flesh from uncontaminated areas varied from 0.170 to 0.214 mg copper per oyster, or from 8.21 to 13.77 mg per 100 grams dry weight (Galtsoff and Whipple 1931,[220] Galtsoff 1964[217]). Oysters growing in adjacent areas slightly contaminated with copper salts had green-colored flesh and contained from 1.27 to 2.46 mg copper per oyster, or from 121 to 271 mg per 100 grams dry weight.

If an effluent containing a variety of components is associated with a tainting problem, identification of the taint-producing component or components is necessary for efficient isolation and removal in waste treatment. For example, Shumway (1966)[228] exposed salmon to various concentrations of wastes and waste components discharged from a plant producing pesticides. Although concentrations of the combined wastes at about 50 to 100 mg/l were found to impart objectionable flavor to test fish, one of the major components of the plant waste, 2,4-dichlorophenol, was found capable of impairing flavor at exposure levels of about 1 to 3 μg/l.

A preliminary laboratory study (English et al. 1963)[214] showed that outboard motor exhaust damages the quality of water in several ways, the most noticeable of which are unpleasant taste and odor in the water and off-flavoring of fish flesh. A later field study (Surber et al. 1965)[230] determined the threshold level of tainting of fish in pond and lake waters to be about 2.6 gal/acre-foot of fuel as exhaust, accumulating over a 2-month period. The gasoline used was regular grade, and the lubricating oil (½ pint/gal) was a popular brand of packaged outboard motor oil.

UPTAKE AND LOSS OF FLAVOR-IMPAIRING MATERIALS

Experiments involving method and rates of uptake and loss of flavor-impairing materials by aquatic organisms have been reported by few investigators. From data available it is obvious that rates are highly variable. Thaysen and Pentelow (1936)[232] exposed trout to extract from odoriferous *Actinomyces*. They showed that fish exposed to 10 ppm of extract acquired an off-flavor in one hour. The exposed fish were also removed and held in uncontaminated water for periods up to five days. The level of tainting, which showed no diminution after 27 hours, became less marked after 2 to 3 days, and no tainting could be detected after 5 days in fresh water.

Shumway and Palensky (*unpublished data*)[239] exposed trout to three separate concentrations of each of the following chemicals, *o*-cresol, 2,4-dichlorophenol, pyridine, and *n*-butylmercaptan, for periods up to 168 hours. With all four chemicals, maximum off-flavor generally occurred in 33.5 hours or less. In a few exceptions, a gradual increase in off-flavor appeared to occur with increasing time up to 168 hours, although the magnitude of increase in off-flavor with time was minor in nature. In tests with *o*-chlorophenol, Boetius (1954)[212] reported that eels required up to 11 days exposure before flavor impairment was detected. The time required to impair flavor was found to be related to the exposure concentration, with low concentrations requiring longer exposure periods.

Shumway (1966)[228] found that the flesh of salmon exposed experimentally to industrial wastes containing mainly phenols acquired maximum off-flavor in 35 hours or less, with much of the tainting occurring within the first 6 hours. After the salmon were transferred to uncontaminated water, most of the acquired off-flavor was lost within 20 hours, although some off-flavor remained up to 72 hours.

In other tests, Shumway and Palensky (*unpublished data*)[239] observed flavor impairment in trout after 24-hour exposure to 2,4-dichlorophenol. After only 33.5 hours in uncontaminated water, the flavor of the trout had returned to the preexposure level, with most of the reduction in off-flavor occurring within 6.5 hours.

Korschgen et al. (1970)[222] transferred carp (*Cyprinus carpio*) to uncontaminated ponds from two sites, one of which received effluents from a major municipality and one of which received little or no effluent. Retention up to 18 days in the holding ponds failed to improve the flavor of the carp from the contaminated site. These authors also reported that channel catfish (*Ictalurus punctatus*) transferred from the Ohio River to control water lost about half of their off-flavor in 7 days and nearly all of it in 21 days.

IDENTIFICATION OF CAUSES OF OFF-FLAVORED ORGANISMS

Determination that a tainting problem exists, or identification of a taint-causing material, involves field or laboratory exposure periods and organoleptic tests. When properly conducted, these tests are reliable but time-consuming. Wright (1966)[235] reported on the use of gas chromatography in conjunction with organoleptic tests. The chromatographic scans were compared with scans of industrial process waste streams to identify the taint-producing wastes. Gas chromatographic techniques are employed routinely in food technology laboratories investigating flavor and odor properties (Rhoades and Millar 1965).[226]

EXPOSURE AND ORGANOLEPTIC TESTS

Field exposure tests (bioassays) are used to determine the existence or the magnitude of a tainting problem in a water body. Fish or other edible aquatic life are held for a period of time in cages at selected locations in and around a suspected problem area or waste discharge and eventually evaluated for flavor. Laboratory bioassays are normally utilized to determine the tainting potential of wastes, waste components, or specific chemicals. Although either static or continuous-flow bioassays can be used in laboratory tests, continuous-flow systems are considered far superior to static tests. Exposure bioassays are followed by the organoleptic evaluation of the flesh of the test organisms.

In their studies of tainted organisms, investigators have used a number of different bioassay and flavor-evaluation procedures, some of which have produced poorly defined results. The following guidelines are based primarily on the successful procedures of Shumway and Newton (*personal communications*).[238]

Test Fish

The flesh of the fish to be exposed should be mild and consistent in flavor. For convenience in holding and taste testing, fish weighing between 200 and 400 grams are desirable, although smaller or larger fish are acceptable. Largemouth bass (*Micropterus salmoides*), yellow perch (*Perca flavescens*), channel catfish, bluegill (*Lepomis macrochirus*), trout, salmon flatfishes (Pleuronectiformes), and others have proven to be acceptable test fish.

Exposure Period

In general, test fish should be exposed for a period not less than 48 hours. Shorter or longer exposures will be advisable in some situations, although possible stress, disease, and mortality resulting from longer retention of test fish and maintenance of holding facilities may negate advantages of long exposure.

Exposure Conditions

The following conditions are desirable in laboratory bioassays:

Dissolved oxygen........	near saturation
Temperature...........	10–15 C for salmonids, and 20–25 C for warmwater fish
pH....................	6.0–8.0, or pH of receiving water
Light..................	intensity held at a low level
Water..................	uncontaminated, or quality of the receiving water; never distilled water

Preparation of Test Fish and Evaluation

Exposed fish and control fish, either fresh or fresh-frozen and subsequently thawed, are individually double-wrapped in aluminum foil, placed in an oven and cooked at about 375 F for 15 to 30 minutes, as size requires. Large fish may be portioned for cooking. No seasoning of any kind is added. Portions of the cooked fish may be placed in small coded cups and served warm to the judges for flavor evaluation. A known "reference" may be provided to aid judges in making comparisons. A minimum of ten experienced judges, each seated in an isolation booth or similar area, smell, taste, and score each sample. This method offers tighter control of variables and conforms more to off-flavor evaluations conducted in food laboratories than the more informal procedure below.

An alternative method is to place the cooked fish, still partially wrapped in foil to preserve the heat and flavor, on a large table. The judges start concurrently and work their way around the table, recording aroma and flavor. If a judge tastes more than six samples during a test, a lessening of organoleptic acuity may occur.

When investigating the potential of a substance to produce taint, a word-evaluation scale for intensity of off-flavor ranging from no off-flavor to extreme off-flavor, has proven successful with trained, experienced judges. Numerical values from 0 to 6 are applied to the word scale for derivation of off-flavor indices and statistical evaluation.

When using the above method, less experienced judges tend to over-react to slight off-flavor. For this reason, in less formal tests evaluating the effect of a substance on the palatability of the organism, an hedonic scale accompanied by word-judgments describing palatability is appropriate, i.e., 0—excellent, 1—very good, 2—good, 3—fair, 4—just acceptable, 5—not quite acceptable, 6—very poor, inedible, and 7—extremely poor, repulsive. Scores of the judges on each sample are averaged to determine final numerical or word-judgment values.

To determine whether there are acceptability differences between controls and test organisms, a triangle test may be used in which two samples are alike and one is different. Judges are asked to select the like samples, to indicate the degree of difference, and to rate both the like and the odd samples on a preference scale.

STATISTICAL EVALUATION

The triangle test is particularly well adapted to statistical analysis, but the organoleptic testing necessary is more extensive than when hedonic scales are used.

Application of the two-way analyses of variance to hedonic-scale data is an acceptable test, but professional assistance with statistical procedures is desirable. Reliance on the word-judgment system is sufficient for general information purposes.

Recommendations

- **To prevent tainting of fish and other edible aquatic organisms, it is recommended that substances which cause tainting should not be present in water in concentrations that lower the acceptability of such organisms as determined by exposure bioassay and organoleptic tests.**
- **Values in Tables III–9 and III–10 are recommended as guidelines in determining what concentrations of wastes and substances in water may cause tainting of the flesh of fish or other aquatic organisms.**

HEAT AND TEMPERATURE

Living organisms do not respond to the quantity of heat but to degrees of temperature or to temperature changes caused by transfer of heat. The importance of temperature to acquatic organisms is well known, and the composition of aquatic communities depends largely on the temperature characteristics of their environment. Organisms have upper and lower thermal tolerance limits, optimum temperatures for growth, preferred temperatures in thermal gradients, and temperature limitations for migration, spawning, and egg incubation. Temperature also affects the physical environment of the aquatic medium, (e.g., viscosity, degree of ice cover, and oxygen capacity. Therefore, the composition of aquatic communities depends largely on temperature characteristics of the environment. In recent years there has been an accelerated demand for cooling waters for power stations that release large quantities of heat, causing, or threatening to cause, either a warming of rivers, lakes, and coastal waters, or a rapid cooling when the artificial sources of heat are abruptly terminated. For these reasons, the environmental consequences of temperature changes must be considered in assessments of water quality requirements of aquatic organisms.

The "natural" temperatures of surface waters of the United States vary from 0 C to over 40 C as a function of latitude, altitude, season, time of day, duration of flow, depth, and many other variables. The agents that affect the natural temperature are so numerous that it is unlikely that two bodies of water, even in the same latitude, would have exactly the same thermal characteristics. Moreover, a single aquatic habitat typically does not have uniform or consistent thermal characteristics. Since all aquatic organisms (with the exception of aquatic mammals and a few large, fast-swimming fish) have body temperatures that conform to the water temperature, these natural variations create conditions that are optimum at times, but are generally above or below optima for particular physiological, behavioral, and competitive functions of the species present.

Because significant temperature changes may affect the composition of an aquatic or wildlife community, an induced change in the thermal characteristics of an ecosystem may be detrimental. On the other hand, altered thermal characteristics may be beneficial, as evidenced in most fish hatchery practices and at other aquacultural facilities. (See the discussion of Aquaculture in Section IV.)

The general difficulty in developing suitable criteria for temperature (which would limit the addition of heat) lies in determining the deviation from "natural" temperature a particular body of water can experience without suffering adverse effects on its biota. Whatever requirements are suggested, a "natural" seasonal cycle must be retained, annual spring and fall changes in temperature must be gradual, and large unnatural day-to-day fluctuations should be avoided. In view of the many variables, it seems obvious that no single temperature requirement can be applied uniformly to continental or large regional areas; the requirements must be closely related to each body of water and to its particular community of organisms, especially the important species found in it. These should include invertebrates, plankton, or other plant and animal life that may be of importance to food chains or otherwise interact with species of direct interest to man. Since thermal requirements of various species differ, the social choice of the species to be protected allows for different "levels of protection" among water bodies as suggested by Doudoroff and Shumway (1970)[272] for dissolved oxygen criteria. (See Dissolved Oxygen, p. 131.) Although such decisions clearly transcend the scientific judgments needed in establishing thermal criteria for protecting selected species, biologists can aid in making them. Some measures useful in assigning levels of importance to species are: (1) high yield to commercial or sport fisheries, (2) large biomass in the existing ecosystem (if desirable), (3) important links in food chains of other species judged important for other reasons, and (4) "endangered" or unique status. If it is desirable to attempt strict preservation of an existing ecosystem, the most sensitive species or life stage may dictate the criteria selected.

Criteria for making recommendations for water temperature to protect desirable aquatic life cannot be simply a maximum allowed change from "natural temperatures." This is principally because a change of even one degree from an ambient temperature has varying significance for an organism, depending upon where the ambient level lies within the tolerance range. In addition, historic temperature records or, alternatively, the existing ambient temperature prior to any thermal alterations by man are not always reliable indicators of desirable conditions for aquatic populations. Multiple developments of water resources also change water temperatures both upward (e.g., upstream power plants or shallow reservoirs) and downward (e.g., deepwater releases from large reservoirs), so that "ambient" and "natural" are exceedingly difficult to define at a given point over periods of several years.

Criteria for temperature should consider both the multiple thermal requirements of aquatic species and requirements for balanced communities. The number of distance requirements and the necessary values for each require periodic reexamination as knowledge of thermal effects on aquatic species and communities increases. Currently definable requirements include:

- maximum sustained temperatures that are consistent with maintaining desirable levels of productivity;
- maximum levels of metabolic acclimation to warm temperatures that will permit return to ambient winter temperatures should artificial sources of heat cease;
- temperature limitations for survival of brief exposures to temperature extremes, both upper and lower;
- restricted temperature ranges for various stages of reproduction, including (for fish) gonad growth and gamete maturation, spawning migration, release of gametes, development of the embryo, commencement of independent feeding (and other activities) by juveniles; and temperatures required for metamorphosis, emergence, and other activities of lower forms;
- thermal limits for diverse compositions of species of aquatic communities, particularly where reduction in diversity creates nuisance growths of certain organisms, or where important food sources or chains are altered;
- thermal requirements of downstream aquatic life where upstream warming of a cold-water source will adversely affect downstream temperature requirements.

Thermal criteria must also be formulated with knowledge of how man alters temperatures, the hydrodynamics of the changes, and how the biota can reasonably be expected to interact with the thermal regimes produced. It is not sufficient, for example, to define only the thermal criteria for sustained production of a species in open waters, because large numbers of organisms may also be exposed to thermal changes by being pumped through the condensers and mixing zone of a power plant. Design engineers need particularly to know the biological limitations to their design options in such instances. Such considerations may reveal nonthermal impacts of cooling processes that may outweigh temperature effects, such as impingement of fish upon intake screens, mechanical or chemical damage to zooplankton in condensers, or effects of altered current patterns on bottom fauna in a discharge area. The environmental situations of aquatic organisms (e.g., where they are, when they are there, in what numbers) must also be understood. Thermal criteria for migratory species should be applied to a certain area only when the species is actually there. Although thermal effects of power stations are currently of great interest, other less dramatic causes of temperature change including deforestation, stream channelization, and impoundment of flowing water must be recognized.

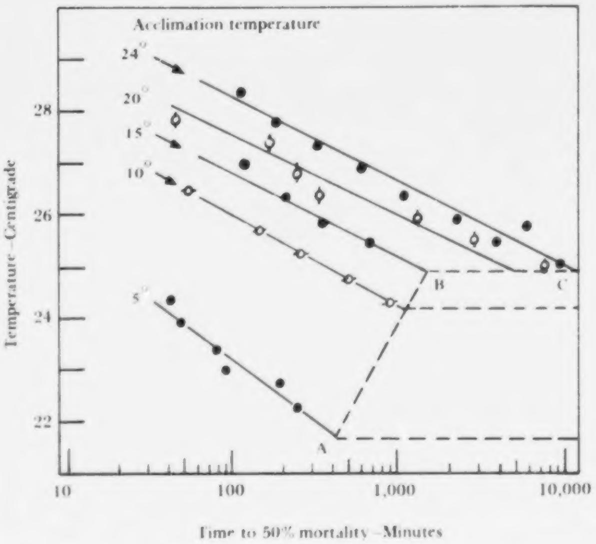
DEVELOPMENT OF CRITERIA

Thermal criteria necessary for the protection of species or communities are discussed separately below. The order of presentation of the different criteria does not imply priority for any one body of water. The descriptions define preferred methods and procedures for judging thermal requirements, and generally do not give numerical values (except in Appendix II–C). Specific values for all limitations would require a biological handbook that is far beyond the scope of this Section. The criteria may seem complex, but they represent an extensively developed framework of knowledge about biological responses. (A sample application of these criteria begins on page 166, Use of Temperature Criteria.)

TERMINOLOGY DEFINED

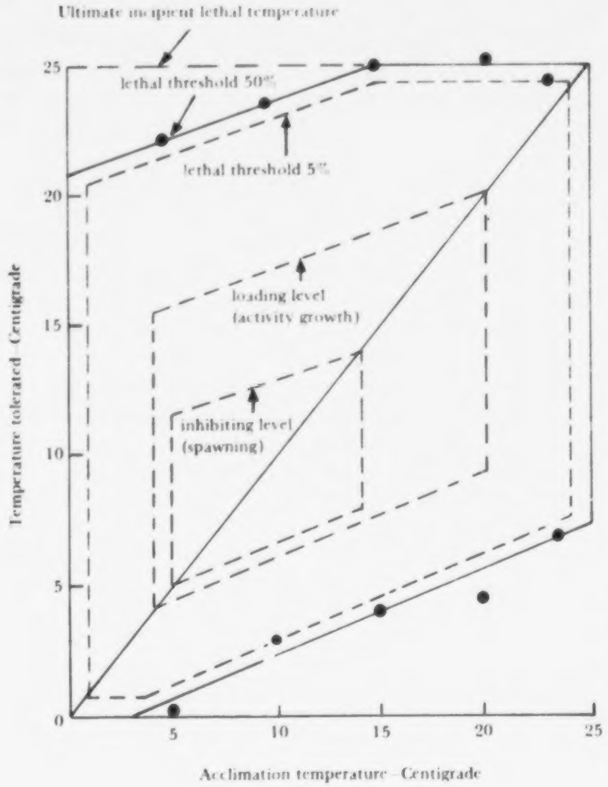
Some basic thermal responses of aquatic organisms will be referred to repeatedly and are defined and reviewed briefly here. Effects of heat on organisms and aquatic communities have been reviewed periodically (e.g., Bullock 1955,[259] Brett 1956;[253] Fry 1947,[276] 1964,[278] 1967;[279] Kinne 1970[296]). Some effects have been analyzed in the context of thermal modification by power plants (Parker and Krenkel 1969;[308] Krenkel and Parker 1969;[298] Cairns 1968;[261] Clark 1969;[263] and Coutant 1970c[269]). Bibliographic information is available from Kennedy and Mihursky (1967),[294] Raney and Menzel (1969),[313] and from annual reviews published by the Water Pollution Control Federation (Coutant 1968,[265] 1969,[266] 1970a,[267] 1971[270]).

Each species (and often each distinct life-stage of a species) has a characteristic tolerance range of temperature as a consequence of acclimations (internal biochemical adjustments) made while at previous holding temperature (Figure III-2; Brett 1956[253]). Ordinarily, the ends of this range, or the lethal thresholds, are defined by survival of 50 per cent of a sample of individuals. Lethal thresholds typically are referred to as "incipient lethal temperatures," and temperature beyond these ranges would be considered "extreme." The tolerance range is adjusted upward by acclimation to warmer water and downward to cooler water, although there is a limit to such accommodation. The lower end of the range usually is at zero degrees centigrade (32 F) for species in temperate latitudes (somewhat less for saline waters), while the upper end terminates in an "ultimate incipient lethal temperature" (Fry et al. 1946[281]). This ultimate threshold temperature represents the "breaking point" between the highest temperatures to which an animal can be acclimated and the lowest of the extreme temperatures that will kill the warm-acclimated organism. Any rate of temperature change over a period of minutes to a few hours will not greatly affect the thermal tolerance limits, since acclimation to changing temperatures requires several days (Brett 1941).[251]



After Brett 1952 [252]

FIGURE III–3—Median resistance times to high temperatures among young chinook* (Oncorhynchus tshawytscha) *acclimated to temperatures indicated. Line A-B denotes rising lethal threshold (incipient lethal temperatures) with increasing acclimation temperature. This rise eventually ceases at the ultimate lethal threshold (ultimate upper incipient lethal temperature), line B-C.



After Brett 1960 [254]

FIGURE III–2—Upper and lower lethal temperatures for young sockeye salmon* (Oncorhynchus nerka) *plotted to show the zone of tolerance. Within this zone two other zones are represented to illustrate (1) an area beyond which growth would be poor to none-at-all under the influence of the loading effect of metabolic demand, and (2) an area beyond which temperature is likely to inhibit normal reproduction.

At the temperatures above and below the incipient lethal temperatures, survival depends not only on the temperature but also on the duration of exposure, with mortality occurring more rapidly the farther the temperature is from the threshold (Figure III–3). (See Coutant 1970a[267] and 1970b[268] for further discussion based on both field and laboratory studies.) Thus, organisms respond to extreme high and low temperatures in a manner similar to the dosage-response pattern which is common to toxicants, pharmaceuticals, and radiation (Bliss 1937).[249] Such tests seldom extend beyond one week in duration.

MAXIMUM ACCEPTABLE TEMPERATURES FOR PROLONGED EXPOSURES

Specific criteria for prolonged exposure (1 week or longer) must be defined for warm and for cold seasons. Additional criteria for gradual temperature (and life cycle) changes during reproduction and development periods are discussed on pp. 162–165.

SPRING, SUMMER, AND FALL MAXIMA FOR PROLONGED EXPOSURE

Occupancy of habitats by most aquatic organisms is often limited within the thermal tolerance range to temperatures somewhat below the ultimate upper incipient lethal temperature. This is the result of poor physiological performance at near lethal levels (e.g., growth, metabolic scope for activities, appetite, food conversion efficiency), interspecies competition, disease, predation, and other subtle ecological factors (Fry 1951;[277] Brett 1971[256]). This complex limitation is evidenced by restricted southern and altitudinal distributions of many species. On the other hand, optimum temperatures (such as those producing fastest growth rates) are not generally necessary at all times to maintain thriving populations and are often exceeded in nature during summer months (Fry 1951;[277] Cooper 1953;[264] Beyerle and Cooper 1960;[246] Kramer and Smith 1960[297]). Moderate temperature fluctuations can generally be tolerated as long as a maximum upper limit is not exceeded for long periods.

A true temperature limit for exposures long enough to reflect metabolic acclimation and optimum ecological performance must lie somewhere between the physiological optimum and the ultimate upper incipient lethal temperatures. Brett (1960)[254] suggested that a provisional long-term exposure limit be the temperature greater than optimum that allowed 75 per cent of optimum performance. His suggestion has not been tested by definitive studies.

Examination of literature on performance, metabolic rate, temperature preference, growth, natural distribution, and tolerance of several species has yielded an apparently sound theoretical basis for estimating an upper temperature limit for long term exposure and a method for doing this with a minimum of additional research. New data will provide refinement, but this method forms a useful guide for the present time. The method is based on the general observations summarized here and in Figure III-4(a, b, c).

1. Performances of organisms over a range of temperatures are available in the scientific literature for a variety of functions. Figures III-4a and b show three characteristic types of responses numbered 1 through 3, of which types 1 and 2 have coinciding optimum peaks. These optimum temperatures are characteristic for a species (or life stage).

2. Degrees of impairment from optimum levels of various performance functions are not uniform with increasing temperature above the optimum for a single species. The most sensitive function appears to be growth rate, for which a temperature of zero growth (with abundant food) can be determined for important species and life stages. Growth rate of organisms appears to be an integrator of all factors acting on an organism. Growth rate should probably be expressed as net biomass gain or net growth (McCormick et al. 1971)[302] of the population, to account for deaths.

3. The maximum temperature at which several species are consistently found in nature (Fry 1951;[277] Narver 1970)[308] lies near the average of the optimum temperature and the temperature of zero net growth.

4. Comparison of patterns in Figures III-4a and b among different species indicates that while the trends are similar, the optimum is closer to the lethal level in some species than it is in sockeye salmon. Invertebrates exhibit a pattern of temperature effects on growth rate that is very similar to that of fish (Figure III-4c).

The optimum temperature may be influenced by rate of feeding. Brett et al. (1969)[257] demonstrated a shift in optimum toward cooler temperatures for sockeye salmon when ration was restricted. In a similar experiment with channel catfish, Andrews and Stickney (1972)[242] could see no such shift. Lack of a general shift in optimum may be due to compensating changes in activity of the fish (Fry *personal observation*).[326]

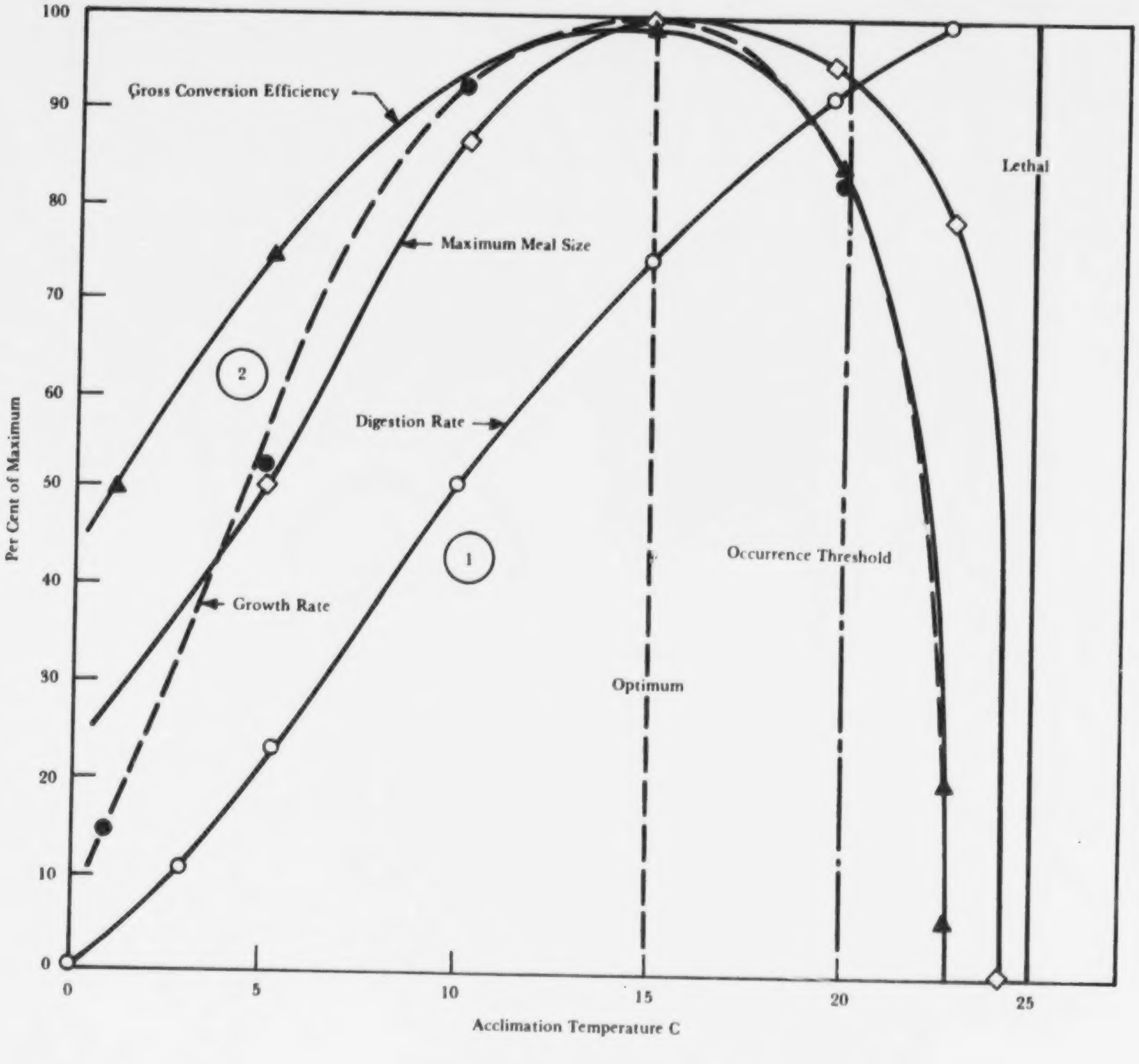
These observations suggest that an average of the optimum temperature and the temperature of zero net growth [(opt. temp. + z.n.g. temp)/2] would be a useful estimate of a limiting weekly mean temperature for resident organisms, providing the peak temperatures do not exceed values recommended for short-term exposures. Optimum growth rate would generally be reduced to no lower than 80 per cent of the maximum if the limiting temperature is as averaged above (Table III-11). This range of reduction from optimum appears acceptable, although there are no quantitative studies available that would allow the criterion to be based upon a specific level of impairment.

The criteria for maximum upper temperature must allow for seasonal changes, because different life stages of many species will have different thermal requirements for the average of their optimum and zero net growths. Thus a juvenile fish in May will be likely to have a lower maximum acceptable temperature than will the same fish in July, and this must be reflected in the thermal criteria for a waterbody.

TABLE III-11—Summary of Some Upper Limiting Temperatures in C, (for periods longer than one week) Based Upon Optimum Temperatures and Temperatures of Zero Net Growth.

Species	Optimum	Zero net growth	Reference	$\frac{\text{opt} + \text{z.n.g.}}{2}$	% of optimum
Catostomus commersoni (white sucker)	27	29.6	*	28.3	86
Coregonus artedii (cisco or lake herring)	16	21.2	McCormick et al. 1971[302]	18.6	82
Ictalurus punctatus (channel catfish)	30	35.7	Strawn 1970[290]	32.8	94
"	30	35.7	Andrews and Stickney 1972[242]	32.8	88
Lepomis macrochirus (bluegill) (year II)	22	28.5	McComish 1971[301]	25.3	82
Micropterus salmoides (largemouth bass)	27.5	34	Strawn 1961[319]	30.8	83
Notropis atherinoides (emerald shiner)	27	33	*	30.5	83
Salvelinus fontinalis (brook trout)	15.4	18.8	*	17.1	80

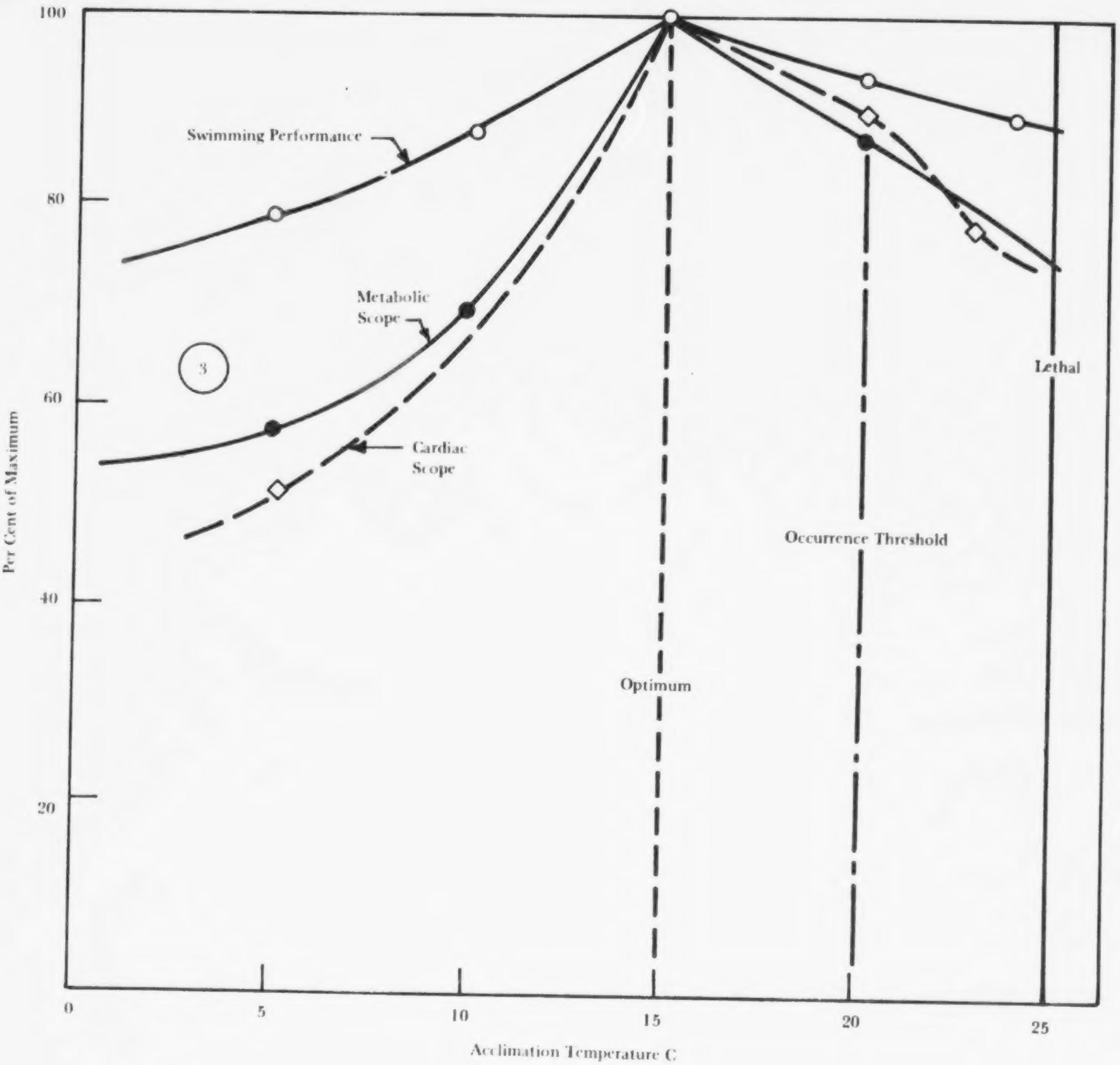
*National Water Quality Laboratory, Duluth, Minn., unpublished data.[318]



After Brett 1971[256]

FIGURE III–4a—Performance of Sockeye Salmon (Oncorhynchus nerka) *in Relation to Acclimation Temperature*

While this approach to developing the maximum sustained temperature appears justified on the basis of available knowledge, few limits can be derived from existing data in the literature on zero growth. On the other hand, there is a sizeable body of data on the ultimate incipient lethal temperature that could serve as a substitute for the data on temperature of zero net growth. A practical consideration in recommending criteria is the time required to conduct



After Brett 1971[256]

FIGURE III-4b—Performance of Sockeye Salmon (Oncorhynchus nerka) *in Relation to Acclimation Temperature*

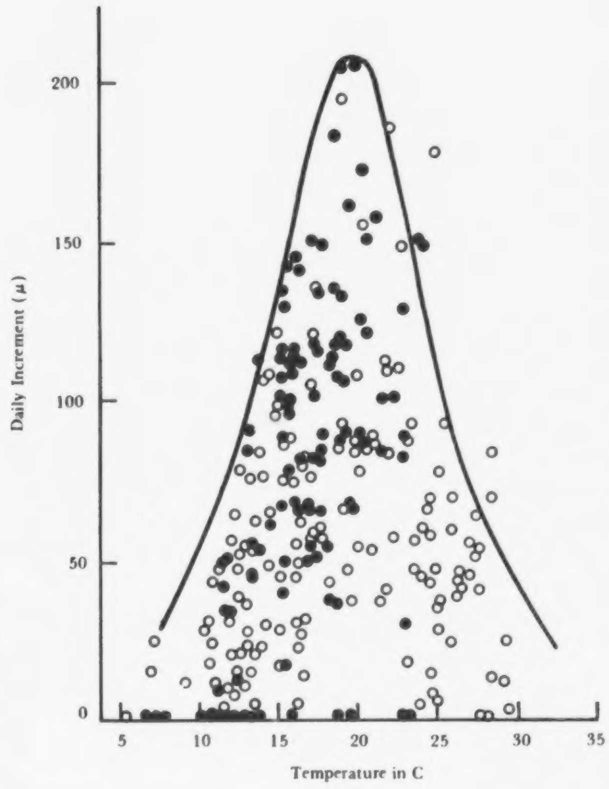
research necessary to provide missing data. Techniques for determining incipient lethal temperatures are standardized (Brett 1952)[252] whereas those for zero growth are not.

A temperature that is one-third of the range between the optimum temperature and the ultimate incipient lethal temperature that can be calculated by the formula

$$\text{optimum temp.} + \frac{\text{ultimate incipient lethal temp.} - \text{optimum temp.}}{3}$$

(*Equation I*)

yields values that are very close to (optimum temp. + z.n.g. temp.)/2. For example, the values are, respectively, 32.7 and 32.8 C for channel catfish and 30.6 and 30.8 for largemouth bass (data from Table III–8 and Appendix II). This formula offers a practical method for obtaining allowable limits, while retaining as its scientific basis the requirements of preserving adequate rates of growth. Some limits obtained from data in the literature are given in Table III–12. A hypothetical example of the effect of this limit on growth of largemouth bass is illustrated in Figure III–5.



Ansell 1968[243]

FIGURE III–4c—*M. mercenaria: *The general relationship between temperature and the rate of shell growth, based on field measurements of growth and temperature.

●: sites in Poole Harbor, England; ○: North American sites.

Figure III–5 shows a hypothetical example of the effects of the limit on maximum weekly average temperature on growth rates of juvenile largemouth bass. Growth data as a function of temperature are from Strawn 1961[319]; the ambient temperature is an averaged curve for Lake Norman, N. C., adapted from data supplied by Duke Power Company. A general temperature elevation of 10 F is used to provide an extreme example. Incremental growth rates (mm/wk) are plotted on the main figure, while annual accumulated growth is plotted in the inset. Simplifying assumptions were that growth rates and the relationship of growth rate to temperature were constant throughout the year, and that there would be sufficient food to sustain maximum attainable growth rates at all times.

The criterion for a specific location would be determined by the most sensitive life stage of an important species likely to be present in that location at that time. Since many fishes have restricted habitats (e.g., specific depth zones) at many life stages, the thermal criterion must be applied to the proper zone. There is field evidence that fish avoid localized areas of unfavorably warm water. This has been demonstrated both in lakes where coldwater fish normally evacuate warm shallows in summer (Smith 1964)[318] and at power station mixing zones (Gammon 1970;[282] Merriman et al. 1965).[304] In most large bodies of water there are both vertical and horizontal thermal gradients that mobile organisms can follow to avoid unfavorable high (or low) temperatures.

The summer maxima need not, therefore, apply to mixing zones that occupy a small percentage of the suitable habitat or necessarily to all zones where organisms have free egress to cooler water. The maxima must apply, however, to restricted local habitats, such as lake hypolimnia or thermoclines, that provide important summer sanctuary areas for cold-water species. Any avoidance of a warm area not part of the normal seasonal habitat of the species will mean that less area of the water body is available to support the population and that production may be reduced. Such reduction should not interfere with biological communities or populations of important species to a degree that is damaging to the ecosystem or other beneficial uses. Nonmobile organisms that must remain in the warm zone will probably be the limiting organisms for that location. Any recommendation for upper limiting temperatures must be applied carefully with understanding of the population dynamics of the species in question in order to establish both local and regional requirements.

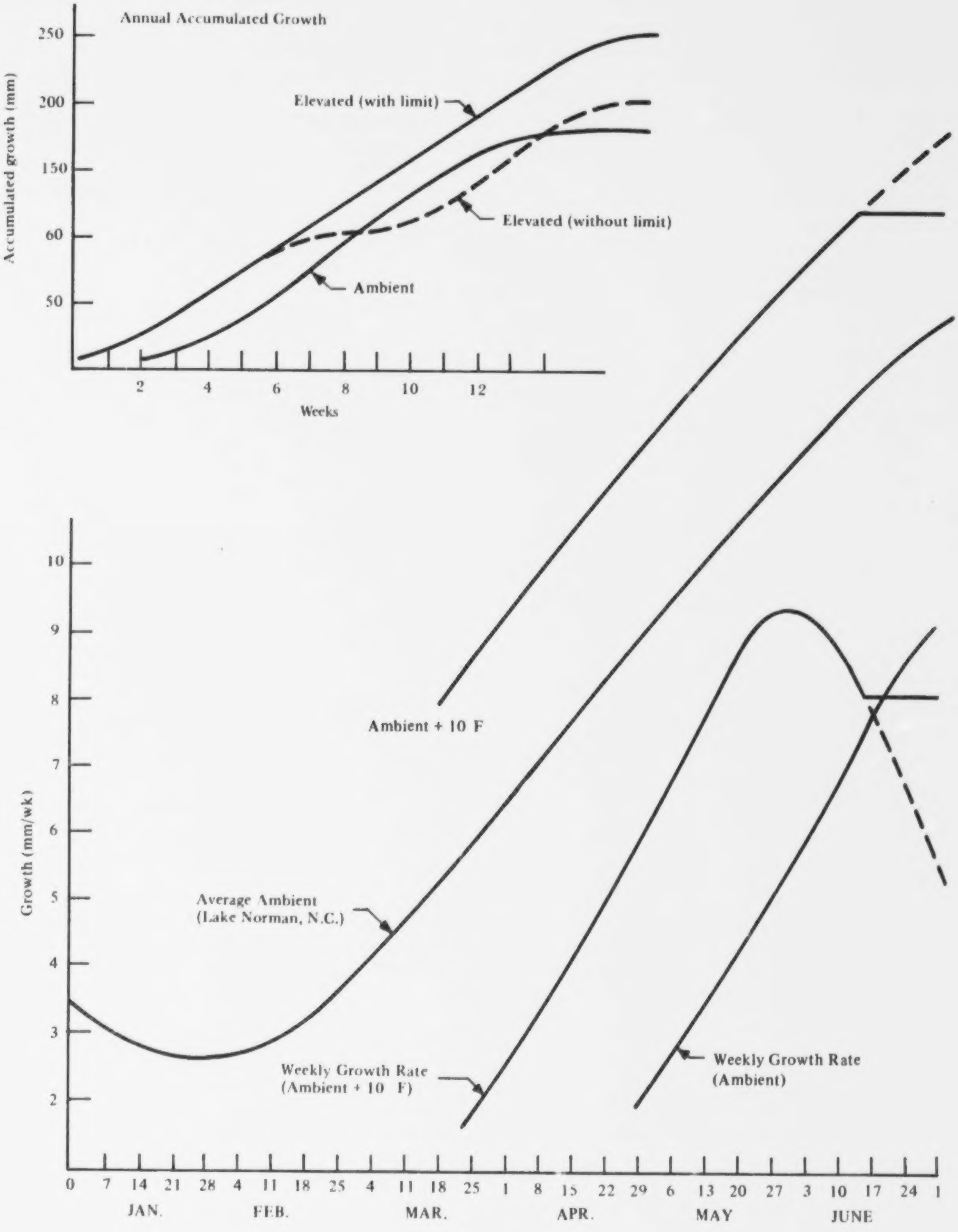

Annual Accumulated Growth
Accumulated growth (mm)
250
200
150
60
50
Elevated (with limit)
Elevated (without limit)
Ambient
2
4
6
8
10
12
Weeks
Growth (mm/wk)
10
9
8
7
6
5
4
3
2
Ambient + 10 F
Average Ambient
(Lake Norman, N.C.)
Weekly Growth Rate
(Ambient + 10 F)
Weekly Growth Rate
(Ambient)
0
7
14
21
28
4
11
18
25
4
11
18
25
1
8
15
22
29
6
13
20
27
3
10
17
24
1
JAN.
FEB.
MAR.
APR.
MAY
JUNE

FIGURE III-5—A hypothetical example of the effects of the limit on maximum weekly average temperature on growth rates of juvenile largemouth bass. Growth data as a function of temperature are from Strawn 1961; the ambient temperature is an averaged curve for Lake Norman, N.C., adapted from data supplied by Duke Power Company. A general temperature elevation of 10 F is used to provide an extreme example. Incremental growth rates (mm/wk) are plotted on the main figure, while annual accumulated growth is plotted in the inset. Simplifying assumptions were that growth rates and the relationship of growth rate to temperature were constant throughout the year, and that there would be sufficient food to sustain maximum attainable growth rates at all times.

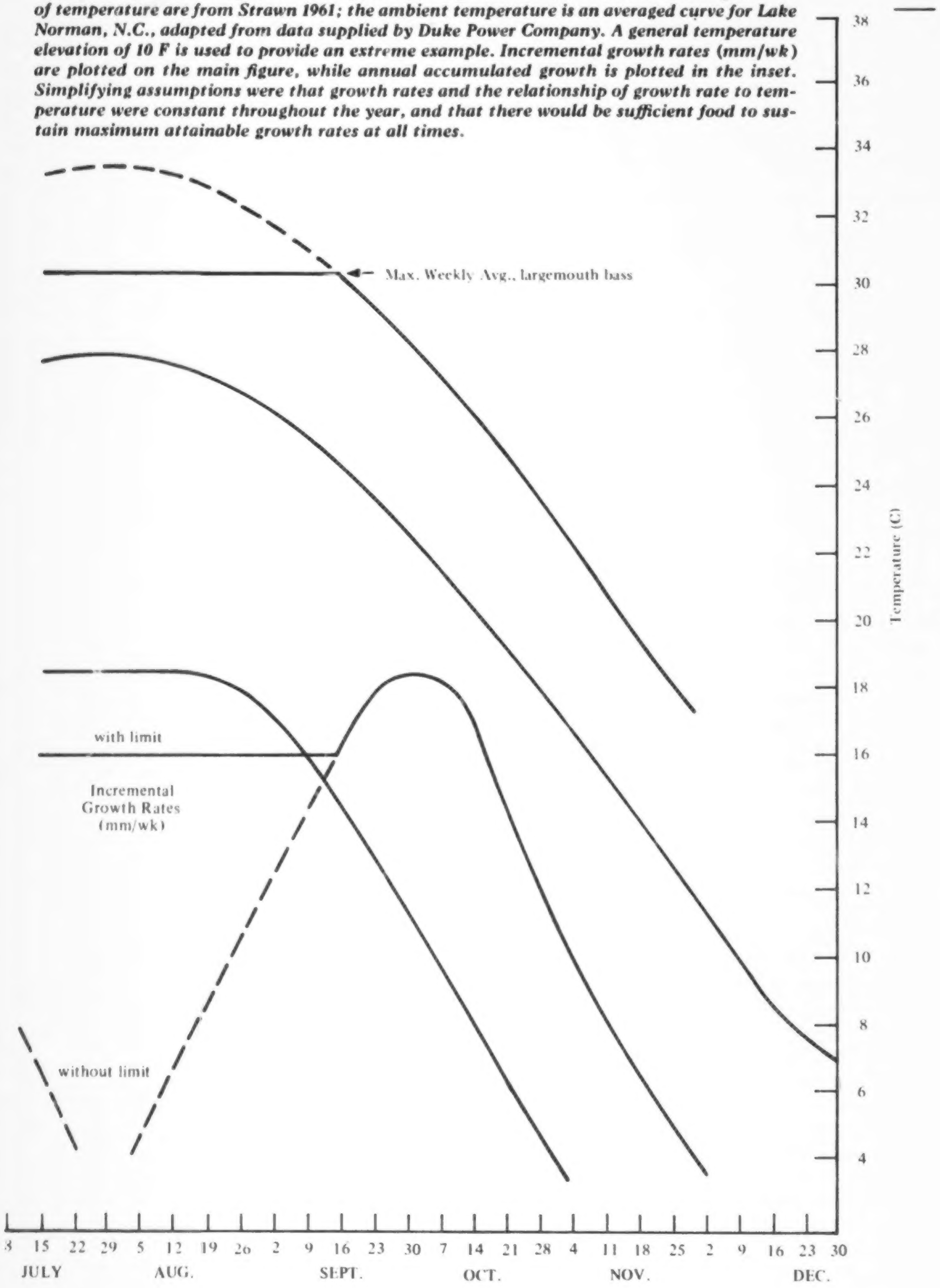


TABLE III-12—Summary of Some Upper Limiting Temperatures for Prolonged Exposures of Fishes Based on Optimum Temperatures and Ultimate Upper Incipient Lethal Temperatures (Equation 1).

Species	Optimum		Function	Reference	Ultimate upper incipient lethal temperature		Reference	Maximum weekly average temperature (Eq 1)	
	C	F			C	F		C	F
Catostomus commersoni (white sucker)	27	80.6	growth	unpubl., NWQL[328]	29.3	84.7	Hart 1947[285]	27.8	82
Coregonus artedii (Cisco or lake herring)	16	60.8	growth	McCormick et al. 1971[302]	25.7	78.3	Edsall and Colby 1970[274]	19.2	66.6
Ictalurus punctatus (channel catfish)	30	86	growth	Strawn 1970;[320] Andrews and Stickney 1971[242]	38.0	100.4	Allen and Strawn 1968[240]	32.7	90.9
Lepomis macrochirus (bluegill) (yr II)	22	71.6	growth	McComish 1971[301] Anderson 1959[241]	33.8	92.8	Hart 1952[286]	25.9	78.6
Micropterus dolomieu (smallmouth bass)	26.3	83	growth	Horning and Pearson 1972[291]	35.0	95.0	Horning and Pearson 1972[291]	29.9	85.8
	28.3	83	growth	Peek 1965[309]					
	ave 27.3	81.1							
Micropterus salmoides (largemouth bass)(fry)	27.5	81.5	growth	Strawn 1961[319]	36.4	97.5	Hart 1952[286]	30.5	86.7
Notropis atherinoides (emerald shiner)	27	80.6	growth	unpubl., NWQL[328]	30.7	87.3	Hart 1952[286]	28.2	82.8
Oncorhynchus nerka (sockeye salmon)	15.0	59.0	growth	Brett et al. 1969[257]	25.0	77.0	Brett 1952[252]	18.3	64.9
	15.0	59.0	other functions	Brett 1971[256]					
(juveniles)	15 0		max. swimming						
Pseudopleuronectes Americanus (winter flounder)	18.0	64.4	growth	Brett 1970[255]	29.1	84.4	Hoff and Westman 1966[289]	21.8	71.2
Salmo trutta (brown trout)	8 to 17	54.5	growth	Brett 1970[255]	23.5	74.3	Bishai 1960[247]	16.2	61.2
	ave 12.5								
Salvelinus fontinalis (brook trout)	15.4	59.7	growth	unpubl, NWQL[328]	25.5	77.9	Fry, Hart and Walker, 1946[281]	18.2	64.8
	13.0	55.4	growth	Baldwin 1957[244]					
	15	59	metabolic	Graham 1949[281]					
	ave 14.5	58.1	scope						
Salvelinus namaycush (lake trout)	16	60.8	scope for activity (2 metabolism)	Gibson and Fry 1954[253]	23.5		Gibson and Fry 1954[253]	18.8	65.8
	17	62.6	swimming speed						
	ave 16.5	61.7							

Heat added to upper reaches of some cold rivers can be retained throughout the river's remaining length (Jaske and Synoground 1970).[292] This factor adds to the natural trend of warming at distances from headwaters. Thermal additions in headwaters, therefore, may contribute substantially to reduction of cold-water species in downstream areas (Mount 1970).[305] Upstream thermal additions should be evaluated for their effects on summer maxima at downstream locations, as well as in the immediate vicinity of the heat source.

Recommendation

Growth of aquatic organisms would be maintained at levels necessary for sustaining actively growing and reproducing populations if the maximum weekly average temperature in the zone inhabited by the species at that time does not exceed one-third of the range between the optimum temperature and the ultimate upper incipient lethal temperature of the species (Equation 1, page 157), and the temperatures above the weekly average do not exceed the criterion for short-term exposures. This maximum need not apply to acceptable mixing zones (see proportional relationships of mixing zones to receiving systems, p. 114), and must be applied with adequate understanding of the normal seasonal distribution of the important species.

WINTER MAXIMA

Although artificially produced temperature elevations during winter months may actually bring the temperature closer to optimum or preferred temperature for important species and attract fish (Trembley 1965),[321] metabolic acclimation to these higher levels can preclude safe return of the organism to ambient temperatures should the artificial heating suddenly cease (Pennsylvania Fish Commission 1971;[310] Robinson 1970)[316] or the organism be driven from the heat area. For example, sockeye salmon (*Oncorhynchus nerka*) acclimated to 20 C suffered 50 percent mortality in the laboratory when their temperature was dropped suddenly to 5 C (Brett 1971;[256] see Figure III-3). The same population of fish withstood a drop to zero when acclimated to 5 C. The lower limit of the range of thermal tolerance of important species must, therefore, be maintained at the normal seasonal ambient temperatures throughout cold seasons, unless special provisions are made to assure that rapid temperature drop will not occur or that organisms cannot become acclimated to elevated temperatures. This can be accomplished by limitations on temperature elevations in such areas as discharge canals and mixing zones where organisms may reside, or by insuring that maximum temperatures occur only in areas not accessible to important aquatic life for lengths of time sufficient to allow metabolic acclimation. Such inaccessible areas would include the high-velocity zones of diffusers or screened discharge channels. This reduction of maximum temperatures would not preclude use of slightly warmed areas as sites for intense winter fisheries.

This consideration may be important in some regions at times other than in winter. The Great Lakes, for example, are susceptible to rapid changes in elevation of the thermocline in summer which may induce rapid decreases in shoreline temperatures. Fish acclimated to exceptionally high temperatures in discharge canals may be killed or severely stressed without changes in power plant operations (Robinson 1968).[314] Such regions should take special note of this possibility.

Some numerical values for acclimation temperatures and lower limits of tolerance ranges (lower incipient lethal temperatures) are given in Appendix II–C. Other data must be provided by further research. There are no adequate data available with which to estimate a safety factor for no stress from cold shocks. Experiments currently in progress, however, suggest that channel catfish fingerlings are more susceptible to predation after being cooled more than 5 to 6 C (Coutant, *unpublished data*).[324]

The effects of limiting ice formation in lakes and rivers should be carefully observed. This aspect of maximum winter temperatures is apparent, although there is insufficient evidence to estimate its importance.

Recommendation

Important species should be protected if the maximum weekly average temperature during winter months in any area to which they have access does not exceed the acclimation temperature (minus a 2 C safety factor) that raises the lower lethal threshold temperature of such species above the normal ambient water temperatures for that season, and the criterion for short-term exposures is not exceeded. This recommendation applies especially to locations where organisms may be attracted from the receiving water and subjected to rapid thermal drop, as in the low velocity areas of water diversions (intake or discharge), canals, and mixing zones.

SHORT-TERM EXPOSURE TO EXTREME TEMPERATURE

To protect aquatic life and yet allow other uses of the water, it is essential to know the lengths of time organisms can survive extreme temperatures (i.e., temperatures that exceed the 7-day incipient lethal temperature). Both natural environments and power plant cooling systems can briefly reach temperature extremes (both upper and lower) without apparent detrimental effect to the aquatic life (Fry 1951;[277] Becker et al. 1971).[245]

The length of time that 50 per cent of a population will survive temperature above the incipient lethal temperature can be calculated from a regression equation of experimental data (such as those in Figure III–3) as follows:

$$\log (\text{time}) = a + b\,(\text{temp.}) \qquad (\textit{Equation 2})$$

where time is expressed in minutes, temperature in degrees centigrade and where a and b are intercept and slope, respectively, which are characteristics of each acclimation temperature for each species. In some cases the time-temperature relationship is more complex than the semi-logarithmic model given above. Equation 2, however, is the most applicable, and is generally accepted by the scientific community (Fry 1967).[279] Caution is recommended in extrapolating beyond the data limits of the original research (Appendix II–C). The rate of temperature change does not appear to alter this equation, as long as the change occurs more rapidly than over several days (Brett 1941;[251] Lemke 1970).[300] Thermal resistance may be diminished by the simultaneous presence of toxicants or other debilitating factors (Ebel et al. 1970,[273] and summary by Coutant 1970c).[269] The most accurate predictability can be derived from data collected using water from the site under evaluation.

Because the equations based on research on thermal tolerance predict 50 per cent mortality, a safety factor is needed to assure no mortality. Several studies have indicated that a 2 C reduction of an upper stress temperature results in no mortalities within an equivalent exposure duration (Fry et al. 1942;[280] Black 1953).[248] The validity of a two degree safety factor was strengthened by the results of Coutant (1970a).[267] He showed that about 15 to 20 per cent of the exposure time, for median mortality at a given high temperature, induced selective predation on thermally shocked salmon and trout. (This also amounted to reduction of the effective stress temperature by about 2 C.) Unpublished data from subsequent predation experiments showed that this reduction of about 2 C also applied to the incipient lethal temperature. The level at which there is no increased vulnerability to predation is the best estimate of a no-stress exposure that is currently available. No similar safety factor has been explored for tolerance of low temperatures. Further research may determine that safety factors, as well as tolerance limits, have to be decided independently for each species, life stage, and water quality situation.

Information needed for predicting survival of a number of species of fish and invertebrates under short-term conditions of heat extremes is presented in Appendix II–C. This information includes (for each acclimation temperature) upper and lower incipient lethal temperatures: coefficients a and b for the thermal resistance equation; and information on size, life stage, and geographic source of the species. It is clear that adequate data are available for only a small percentage of aquatic species, and additional research is necessary. Thermal resistance information should be obtained locally for critical areas to account for simultaneous presence of toxicants or other debilitating factors, a consideration not reflected in Appendix II-C data. More data are available for upper lethal temperatures than for lower.

The resistance time equation, Equation 2, can be rearranged to incorporate the 2 C margin of safety and also to define conditions for survival (right side of the equation less than or equal to 1) as follows:

$$1 \geq \frac{\text{time}}{10^{[a+b(\text{temp.}+2)]}} \qquad (\textit{Equation 3})$$

Low levels of mortality of some aquatic organisms are not necessarily detrimental to ecosystems, because permissible mortality levels can be established. This is how fishing or shellfishing activities are managed. Many states and international agencies have established elaborate systems for setting an allowable rate of mortality (for sport and commercial fish) in order to assure needed reproduction and survival. (This should not imply, however, that a form of pollution should be allowed to take the entire harvestable yield.) Warm discharge water from a power plant may sufficiently stimulate reproduction of some organisms (e.g., zooplankton), such that those killed during passage through the maximally heated areas are replaced within a few hours, and no impact of the mortalities can be found in the open water (Churchill and Wojtalik 1969;[262] Heinle 1969).[288] On the other hand, Jensen (1971)[293] calculated that even five percent additional mortality of 0-age brook trout (*Salvelinus fontinalis*) decreased the yield of the trout fishery, and 50 per cent additional mortality would, theoretically, cause extinction of the population. Obviously, there can be no adequate generalization concerning the impact of short-term effects on entire ecosystems, for each case will be somewhat different. Future research must be directed toward determining the effects of local temperature stresses on population dynamics. A complete discussion will not be attempted here. Criteria for complete short-term protection may not always be necessary and should be applied with an adequate understanding of local conditions.

Recommendation

Unless there is justifiable reason to believe it unnecessary for maintenance of populations of a species, the right side of Equation 3 for that species should not be allowed to increase above unity when the temperature exceeds the incipient lethal temperature minus 2 C:

$$1 \geq \frac{\textbf{time}}{\mathbf{10}^{[a+b(\text{temp.}+2)]}}$$

Values for *a* and *b* at the appropriate acclimation temperature for some species can be obtained from Appendix II–C or through additional research if necessary data are not available. This recommendation applies to all locations where organisms to be protected are exposed, including areas within mixing zones and water diversions such as power station cooling water.

REPRODUCTION AND DEVELOPMENT

The sequence of events relating to gonad growth and gamete maturation, spawning migration, release of gametes, development of the egg and embryo, and commencement of independent feeding represents one of the most complex phenomena in nature, both for fish (Brett 1970)[255] and invertebrates (Kinne 1970).[296] These events are generally the most thermally sensitive of all life stages. Other environmental factors, such as light and salinity, often seasonal in nature, can also profoundly affect the response to temperature (Wiebe 1968).[323] The general physiological state of the organisms (e.g., energy reserves), which is an integration of previous history, has a strong effect on reproductive potential (Kinne 1970).[296] The erratic sequence of failures and successes of different year classes of lake fish attests to the unreliability of natural conditions for providing optimum reproduction.

Abnormal, short-term temperature fluctuations appear to be of greatest significance in reduced production of juvenile fish and invertebrates (Kinne, 1963).[295] Such thermal fluctuations can be a prominent consequence of water use as in hydroelectric power (rapid changes in river flow rates), thermal electric power (thermal discharges at fluctuating power levels), navigation (irregular lock releases), and irrigation (irregular water diversions and wasteway releases). Jaske and Synoground (1970)[292] have documented such temperature changes due to interacting thermal and hydroelectric discharges on the Columbia River.

Tolerable limits or variations of temperature change throughout development, and particularly at the most sensitive life stages, differ among species. There is no adequate summary of data on such thermal requirements for successful reproduction. The data are scattered through many years of natural history observations (however, see Breder and Rosen 1966[250] for a recent compilation of some data; also see Table III-13). High priority must be assigned to summarizing existing information and obtaining that which is lacking.

Uniform elevations of temperature by a few degrees during the spawning period, while maintaining short-term temperature cycles and seasonal thermal patterns, appear to have little overall effect on the reproductive cycle of resident aquatic species, other than to advance the timing for spring spawners or delay it for fall spawners. Such shifts are often seen in nature, although no quantitative measurements of reproductive success have been made in this connection. For example, thriving populations of many fishes occur in diverse streams of the Tennessee Valley in which the date of the spawning temperature may vary in a

TABLE III-13—Spawning Requirements of Some Fish, Arranged in Ascending Order of Spawning Temperatures (Adapted from Wojtalik, T. A., unpublished manuscript)*

Fishes	Temp. (C)	Spawning site	Range in spawning depth	Daily spawning time	Egg site	Incubation period days (Temp. C)
Sauger Stizostedion canadense	5.0	Shallow gravel bars	2-4 feet	Night	Bottom	25 (5.0)
Walleye S. vitreum vitreum	7.0	Gravel, rubble, boulders on bar	3-10 feet	Day, night	Bottom	....
Longnose gar Lepisosteus osseus	10.0	Flooded shallows	Flooded shallows	Day	Weeds	6 (20.0)
White bass Morone chrysops	11.7	Sand & rock shores	3-12 feet	Day, long but esp. night	Surface	2 (15.0)
Least darter Etheostoma microperca	12.0					
Spotted sucker Minytrema melanops	12.8					
White sucker Catostomus commersoni	12.0-13.0	Streams or bars	....	Day, night	Bottom	....
Silvery minnow Hybognathus nuchalis	13.0	Coves	....	Day	Bottom	....
Banded pygmo sunfish Elassoma zonatum	13.9-16.7					
White crappie Pomoxis annularis	14.0-16.0	Submerged materials in shallows	....	Day	Bottom	1 (21.1-23.2)
Fathead minnow Pimephales promelas	14.4 25.0	Shallows	Nr. surface	Day	Underside floating objects	....
Bigmouth buffalo Ictiobus cyprinellus	15.6-18.3	Shallows	....	Day	Bottom	9-10 (18.7)
Largemouth bass Micropterus salmoides	15.5	Shallows near bank	30 inches	Day	Bottom	5 (18.9)
Common shiner Notropis cornutus	15.6-18.3	Small gravel streams	....	Day	Bottom	
Golden shiner Notemigonus crysoleucas	15.6	Bays & shoals, weeds	....	Day	Weeds	4 (15.6+)
Green sunfish Lepomis cyanellus	15.6	Bank, shallows	Inches to 1½ feet	Day	Bottom	....
Paddlefish Polyodon spathula	16.0	Over gravel bars	Nr. surface	Night, day	Bottom	....
Blackside darter Percina maculata	16.5					
Gizzard shad Dorosoma cepedianum	16.7					
Smallmouth bass Micropterus dolomieui	16.7	Gravel rock shore	3-20 feet	Day	Bottom	7 (15.0)
Spotted bass Micropterus punctulatus	17.8	Small streams, bar	....	Day	Bottom	4-5 (20.0)
Johnny darter Etheostoma nigrum	18.0					
Orange spotted sunfish Lepomis humilis	18.3					
Smallmouth buffalo Ictiobus bubalus	18.9					
Black buffalo I. niger	18.9					
Carp Cyprinus carpio	19.0	Flooded shallows	Nr. surface	Day, night	Bottom	4-8 (16.7)
Bluegill Lepomis macrochirus	19.4	Weeds, shallows	2-6 feet	Day	Bottom	1½-3 (22.2)
Redbreast sunfish L. auritis	20.0					
Channel catfish Ictalurus punctatus	20.0 26.7	Bank cavity	<10 feet	Day, night	Bottom	9-10 (15.0)
White catfish I. catus	20.0	Sand gravel bar	<10 feet	Day	Bottom	5-7 (23.9-29.4)
Pumpkinseed Lepomis gibbosus	20.0	Bank shallows	<5 feet	Day	Bottom	3 (27.8)
Black crappie Pomoxis nigromaculatus	20.0					
Brook silverside Labidesthes sicculus	20.0	Over gravel	Surface	Day	Weeds, bottom	....
Brown bullhead Ictalurus nebulosus	21.1	Shallows, weeds	Inches to 6 feet	....	Weeds, bottom	5 (25.0)
Threadfin shad Dorosoma petenense	21.1	Shallow and open water	Surface	Day	Bottom	3 (26.7)
Warmouth Lepomis gulosus	21.0	Bank shallows	<5 feet	Day	Bottom	1½ (25.0-26.7)
River redhorse Moxostoma carinatum	21.7-24.4	Riffles, streams	....	Day	Bottom	....

TABLE III-13—Spawning Requirements of Some Fish, Arranged in Ascending Order of Spawning Temperatures—Continued

Fishes	Temp. (C)	Spawning site	Range in spawning depth	Daily spawning time	Egg site	Incubation period days (Temp. C)
Blue catfish Ictalurus furcatus	22.2					
Flathead catfish Pylodictis olivaris	22.2					
Redear sunfish Lepomis microlophus	23.0	Quiet, various	Inches to 10 feet			
Longear sunfish L. megalotis	23.3					
Freshwater drum Aplodinotus grunniens	23.8					
River carpsucker Carpiodes carpio	23.9					
Spotted bullhead Ictalurus serracanthus	26.7					
Yellow bullhead I. natalis		Quiet, shallows	1½–4 feet		Bottom	5–10 (18.9)

* T. A. Wojtalik, Tennessee Valley Authority, Muscle Shoals, Alabama.[125]

given year by 22 to 65 days. Examination of the literature shows that shifts in spawning dates by nearly one month are common in natural waters throughout the U.S. Populations of some species at the southern limits of their distribution are exceptions, e.g., the lake whitefish (*Coregonus clupeaformis*) in Lake Erie that require a prolonged, cold incubation period (Lawler 1965)[299] and species such as yellow perch (*Perca flavescens*) that require a long chill period for egg maturation prior to spawning (Jones, *unpublished data*).[327]

This biological plasticity suggests that the annual spring rise, or fall drop, in temperature might safely be advanced (or delayed) by nearly one month in many regions, as long as the thermal requirements that are necessary for migration, spawning, and other activities are not eliminated and the necessary chill periods, maturation times, or incubation periods are preserved for important species. Production of food organisms may advance in a similar way, with little disruption of food chains, although there is little evidence to support this assumption (but see Coutant 1968;[265] Coutant and Steele 1968;[271] and Nebeker 1971).[307] The process is similar to the latitudinal differences within the range of a given species.

Highly mobile species that depend upon temperature synchrony among widely different regions or environments for various phases of the reproductive or rearing cycle (e.g., anadromous salmonids or aquatic insects) could be faced with dangers of dis-synchrony if one area is warmed, but another is not. Poor long-term success of one year class of Fraser River (British Columbia) sockeye salmon (*Oncorhynchus nerka*) was attributed to early (and highly successful) fry production and emigration during an abnormally warm summer followed by unsuccessful, premature feeding activity in the cold and still unproductive estuary (Vernon 1958).[322] Anadromous species are able, in some cases, (see studies of eulachon (*Thaleichthys pacificus*) by Smith and Saalfeld 1955)[317] to modify their migrations and spawning to coincide with the proper temperatures whenever and wherever they occur.

Rates of embryonic development that could lead to premature hatching are determined by temperatures of the microhabitat of the embryo. Temperatures of the microhabitat may be quite different from those of the remainder of the waterbody. For example, a thermal effluent at the temperature of maximum water density (approximately 4 C) can sink in a lake whose surface water temperature is colder (Hoglund and Spigarelli, 1972).[290] Incubating eggs of such species as lake trout (*Salvelinus namaycush*) and various coregonids on the lake bottom may be intermittently exposed to temperatures warmer than normal. Hatching may be advanced to dates that are too early for survival of the fry in their nursery areas. Hoglund and Spigarelli 1972,[290] using temperature data from a sinking plume in Lake Michigan, theorized that if lake herring (*Coregonus artedii*) eggs had been incubated at the location of one of their temperature sensors, the fry would have hatched seven days early. Thermal limitations must, therefore, apply at the proper location for the particular species or life stage to be protected.

Recommendations

After their specific limiting temperatures and exposure times have been determined by studies tailored to local conditions, the reproductive activity of selected species will be protected in areas where:

- **periods required for gonad growth and gamete maturation are preserved;**
- **no temperature differentials are created that block spawning migrations, although some delay or advancement of timing based upon local conditions may be tolerated;**

- **temperatures are not raised to a level at which necessary spawning or incubation temperatures of winter-spawning species cannot occur;**
- **sharp temperature changes are not induced in spawning areas, either in mixing zones or in mixed water bodies (the thermal and geographic limits to such changes will be dependent upon local requirements of species, including the spawning microhabitat, e.g., bottom gravels, littoral zone, and surface strata);**
- **timing of reproductive events is not altered to the extent that synchrony is broken where reproduction or rearing of certain life stages is shown to be dependent upon cyclic food sources or other factors at remote locations.**
- **normal patterns of gradual temperature changes throughout the year are maintained.**

These requirements should supersede all others during times when they apply.

CHANGES IN STRUCTURE OF AQUATIC COMMUNITIES

Significant change in temperature or in thermal patterns over a period of time may cause some change in the composition of aquatic communities (i.e., the species represented and the numbers of individuals in each species). This has been documented by field studies at power plants (Trembley 1956 1960)[321] and by laboratory investigations (McIntyre 1968).[303] Allowing temperature changes to alter significantly the community structure in natural waters may be detrimental, even though species of direct importance to man are not eliminated.

The limits of allowable change in species diversity due to temperature changes should not differ from those applicable to any other pollutant. This general topic is treated in detail in reviews by others (Brookhaven National Lab. 1969)[265] and is discussed in Appendix II B, Community Structure and Diversity Indices, p. 408.

NUISANCE ORGANISMS

Alteration of aquatic communities by the addition of heat may occasionally result in growths of nuisance organisms provided that other environmental conditions essential to such growths (e.g., nutrients) exist. Poltoracka (1968)[311] documented the growth stimulation of plankton in an artificially heated small lake; Trembley (1965[321]) reported dense growths of attached algae in the discharge canal and shallow discharge plume of a power station (where the algae broke loose periodically releasing decomposing organic matter to the receiving water). Other instances of algal growths in effluent channels of power stations were reviewed by Coutant (1970c).[269]

Changed thermal patterns (e.g., in stratified lakes) may greatly alter the seasonal appearances of nuisance algal growths even though the temperature changes are induced by altered circulation patterns (e.g., artificial destratification). Dense growths of plankton have been retarded in some instances and stimulated in others (Fast 1968;[275] and *unpublished data* 1971).[325]

Data on temperature limits or thermal distributions in which nuisance growths will be produced are not presently available due in part to the complex interactions with other growth stimulants. There is not sufficient evidence to say that any temperature increase will necessarily result in increased nuisance organisms. Careful evaluation of local conditions is required for any reasonable prediction of effect.

Recommendation

Nuisance growths of organisms may develop where there are increases in temperature or alterations of the temporal or spatial distribution of heat in water. There should be careful evaluation of all factors contributing to nuisance growths at any site before establishment of thermal limits based upon this response, and temperature limits should be set in conjunction with restrictions on other factors (see the discussion of Eutrophication and Nutrients in Section I).

CONCLUSIONS

Recommendations for temperature limits to protect aquatic life consist of the following two upper limits for any time of the year (Figure III-6).

1. One limit consists of a maximum weekly average temperature that:

   (a) in the warmer months (e.g., April through October in the North, and March through November in the South) is one third of the range between the optimum temperature and the ultimate upper incipient lethal temperature for the most sensitive important species (or appropriate life stage) that is normally found at that location at that time; or

   (b) in the cooler months (e.g., mid-October to mid-April in the North, and December to February in the South) is that elevated temperature from which important species die when that elevated temperature is suddenly dropped to the normal ambient temperature, with the limit being the acclimation temperature (minus a 2 C safety factor), when the lower incipient lethal temperature equals the normal ambient water temperature (in some regions this limit may also be applicable in summer); or

   (c) during reproduction seasons (generally April-June and September-October in the North, and March-May and October-November in the South) is that temperature that meets specific site requirements for successful migration, spawning, egg incubation, fry rearing, and other reproductive functions of important species; or

(d) at a specific site is found necessary to preserve normal species diversity or prevent undesirable growths of nuisance organisms.

2. The second limit is the time-dependent maximum temperature for short exposures as given by the species-specific equation:

$$1 \geq \frac{\text{time}}{10^{[a+b(\text{temp.}+2)]}}$$

Local requirements for reproduction should supersede all other requirements when they are applicable. Detailed ecological analysis of both natural and man-modified aquatic environments is necessary to ascertain when these requirements should apply.

USE OF TEMPERATURE CRITERIA

A hypothetical electric power station using lake water for cooling is illustrated as a typical example in Figure III–7. This discussion concerns the application of thermal criteria to this typical situation.

The size of the power station is 1,000 megawatts electric (MW$_e$) if nuclear, or 1,700 MW$_e$ if fossil-fueled (oil, coal, gas); and it releases 6.8 billion British Thermal Units (BTU) per hour to the aquatic environment. This size is representative of power stations currently being installed. Temperature rise at the condensers would be 20 F with cooling water flowing at the rate of 1,520 cubic feet/second (ft^3/sec) or 682,000 gallons/minute. Flow could be increased to reduce temperature rise.

The schematic Figure III–7 is drawn with two alternative discharge arrangements to illustrate the extent to which design features affect thermal impacts upon aquatic life

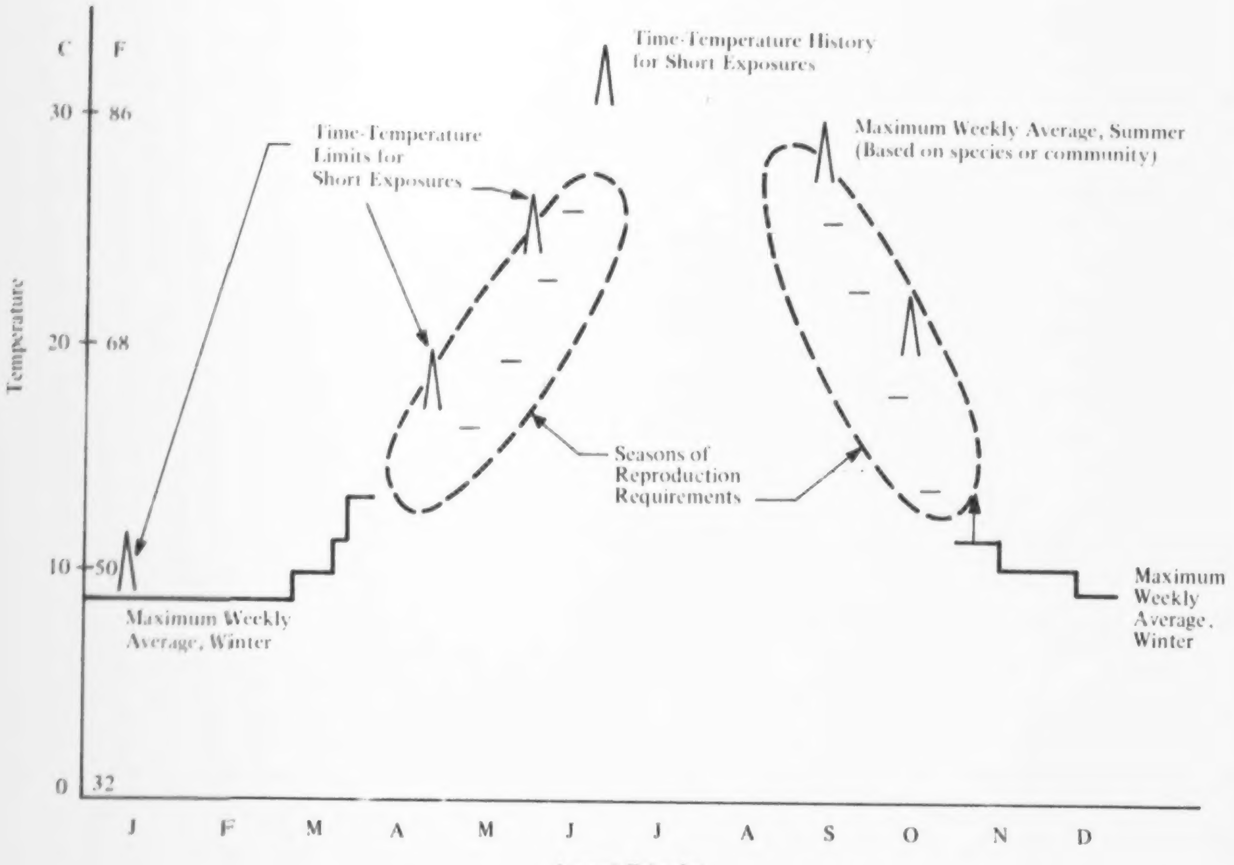


FIGURE III–6—Schematic Summary of Thermal Criteria

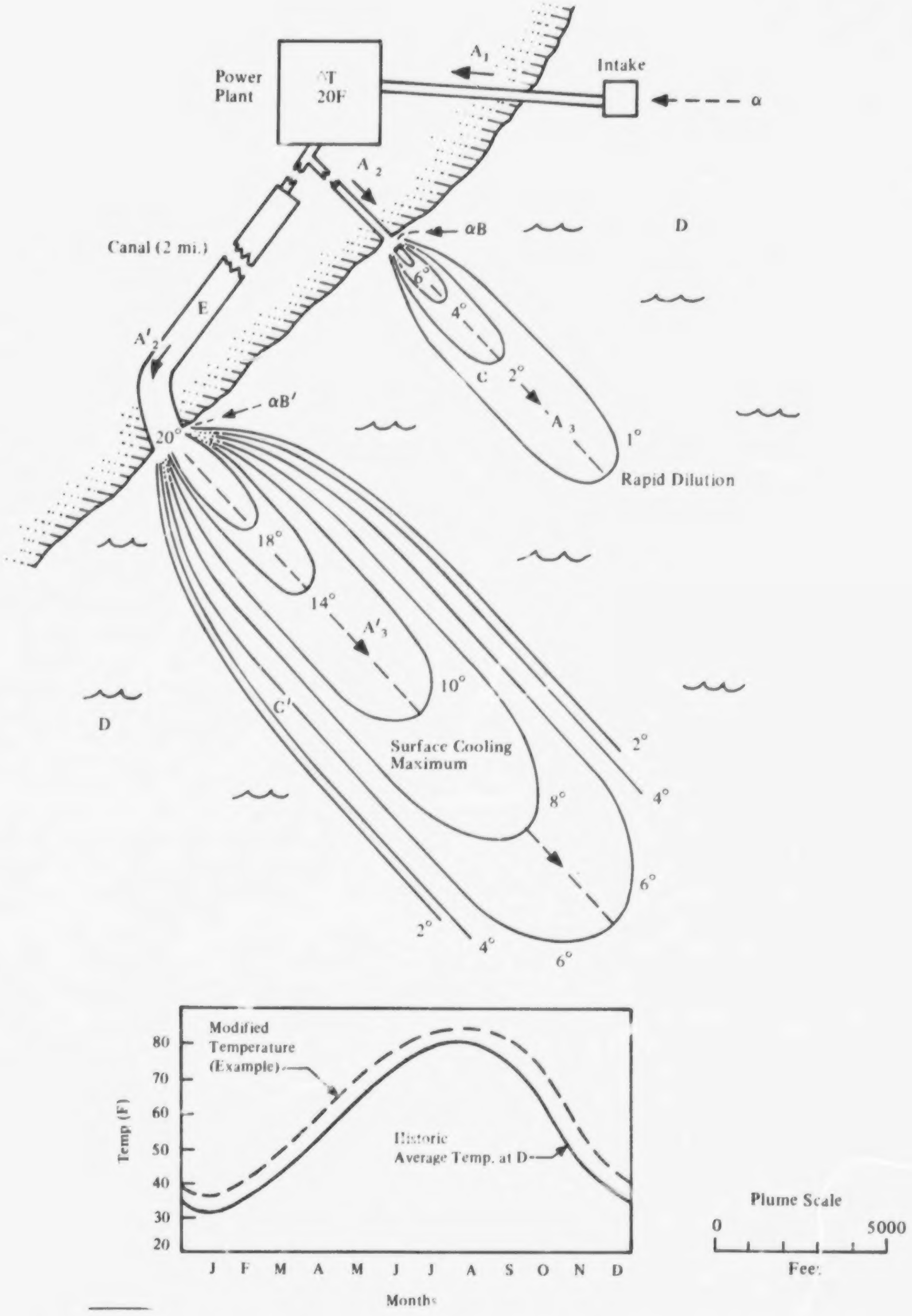

Power Plant
ΔT 20F
A₁
Intake
α
A₂
αB
D
Canal (2 mi.)
E
6°
4°
A′₂
C
2°
A₃
1°
Rapid Dilution
αB′
20°
18°
14°
A′₃
10°
D
C′
Surface Cooling Maximum
2°
4°
8°
6°
2°
4°
6°
Modified Temperature (Example)
Historic Average Temp. at D
Temp (F)
80
70
60
50
40
30
20
J F M A M J J A S O N D
Months
Plume Scale
0
5000
Feet

FIGURE III-7—Hypothetical Power Plant Site For Application of Water Temperature Criteria

Warm condenser water can be carried from the station to the lake by (a) a pipe carrying water at a high flow velocity or (b) a canal in which the warm water flows slowly. There is little cooling in a canal, as measurements at several existing power stations have shown. Water can be released to the lake by using any of several combinations of water velocity and volume (i.e., number of outlets) or outlet dimensions and locations. These design features largely determine the configuration of the thermal plumes illustrated in Figure III–7 resulting from either rapid dilution with lake water or from slow release as a surface layer. The isotherms were placed according to computer simulation of thermal discharges (Pritchard 1971)[312] and represent a condition without lake currents to aid mixing.

Exact configuration of an actual plume depends upon many factors (some of which change seasonally or even hourly) such as local patterns of currents, wind, and bottom and shore topography.

Analytical Steps

Perspective of the organisms in the water body and of the pertinent non-biological considerations (chemical, hydrological, hydraulic) is an essential beginning. This perspective requires a certain amount of literature survey or on site study if the information is not well known. Two steps are particularly important:

1. identification of the important species and community (primary production, species diversity, etc.) that are relevant to this site; and
2. determination of life patterns of the important species (seasonal distribution, migrations, spawning areas, nursery and rearing areas, sites of commercial or sport fisheries). This information should include as much specific information on thermal requirements as it is possible to obtain from the literature.

Other steps relate the life patterns and environmental requirements of the biota to the sources of potential thermal damage from the power plant. These steps can be identified with specific areas in Figure III–7.

Aquatic Areas Sensitive to Temperature Change

Five principal areas offer potential for biological damage from thermal changes, labeled A-E on Figure III–7. (There are other areas associated with mechanical or chemical effects that cannot be treated here; see the index.)

Area A	The cooling water as it passes through the intake, intake piping (A_1), condensers, discharge piping (A_2) or canal (A'_2), and thermal plume (A_3 or A'_3), carrying with it small organisms (such as phytoplankton, zooplankton, invertebrate larvae, and fish eggs or larvae). Organisms receive a thermal shock to the full 20 F above ambient temperature with a duration that depends upon the rate of water flow and the temperature drop in the plume.
Area B	Water of the plume alone that entrains both small and larger organisms (including small fish) as it is diluted (B or B′). Organisms receive thermal shocks from temperatures ranging from the discharge to the ambient temperature, depending upon where they are entrained.
Area C	Benthic environment where bottom organisms (including fish eggs) can be heated chronically or periodically by the thermal plume (C or C′).
Area D	The slightly warmed mixed water body (or large segment of it) where all organisms experience a slightly warmer average temperature (D).
Area E	The discharge canal in which resident or seasonal populations reside at abnormally high temperatures (E).

Cooling Water Entrainment

It is not adequate to consider only thermal criteria for water bodies alone when large numbers of aquatic organisms may be pumped through a power plant. The probability of an organism being pumped through will depend upon the ratio of the volume of cooling water in the plant to the volume in the lake (or to the volume passing the plant in a river or tidal fresh water). Tidal environments (both freshwater and saline) offer greater potential for entrainment than is apparent, since the same water mass will move back and forth past the plant many times during the lifetime of pelagic residence time of most organisms. Thermal shocks that could be experienced by organisms entrained at the hypothetical power station are shown in Figure III–8.

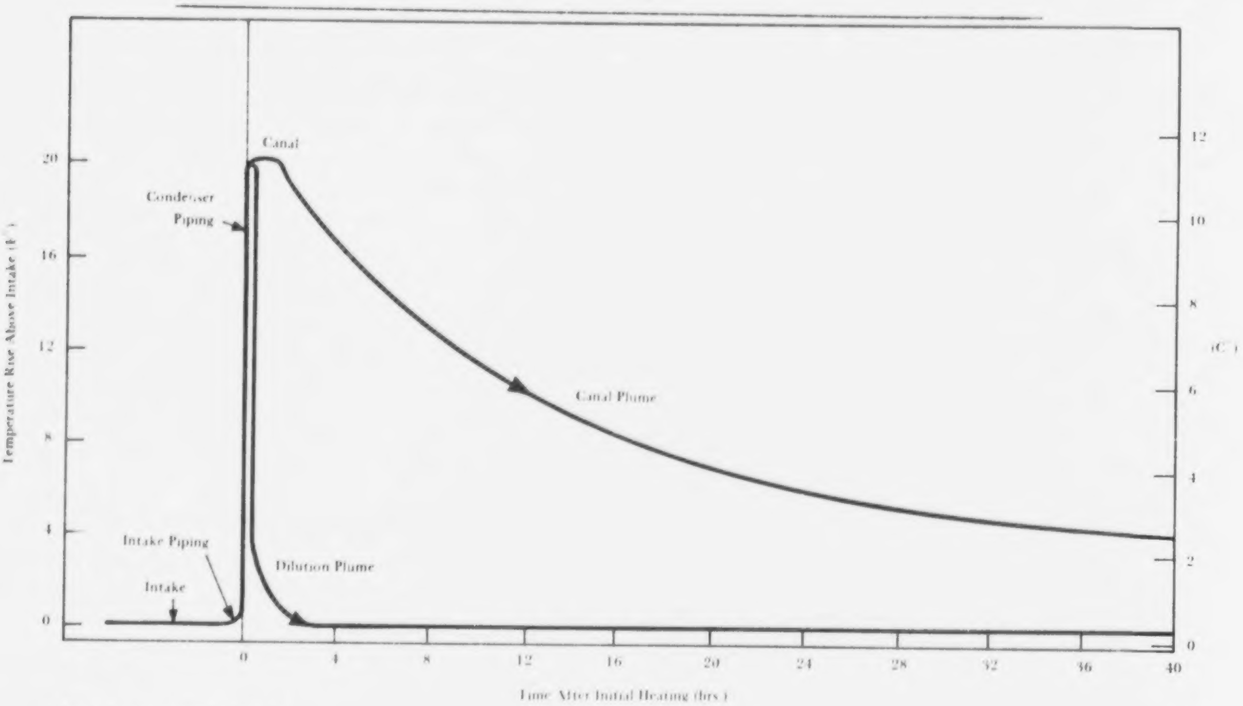
Detrimental effects of thermal exposures received during entrainment can be judged by using the following equation for short-term exposures to extreme temperatures:

$$\text{General criterion: } 1 \geq \frac{\text{time}}{10^{[a+b(\text{temp.}+2)]}}$$

Values for a and b in the equation for the species of aquatic organisms that are likely to be pumped with cooling water may be obtained from Appendix II, or the data may be obtained using the methods of Brett (1952).[252] The prevailing intake temperature would determine the acclimation temperature to be selected from the table.

For example, juvenile largemouth bass may frequent the near-shore waters of this lake and be drawn into the intake. To determine whether the hypothetical thermal discharges (Figure III–7) would be detrimental for juvenile bass, the following analysis can be made (assuming, for example, that the lake is in Wisconsin where these basic data for bass are available):

Criterion for juvenile bass (Wisconsin) when intake



Modified after Coutant 1970;[269]

FIGURE III-8—Time Course of Temperature Change in Cooling Water Passing Through the Example Power Station with Two Alternate Discharges. The Canal Is Assumed to Flow at a Rate of 3 Ft. Per Sec.

temperature (acclimation) is 70 F (21.11 C). (Data from Appendix II-C).

$$1 \geq \frac{\text{time}}{10^{[34.3649-0.9789(\text{temp.}+2)]}}$$

Canal

Criterion applied to entrainment to end of discharge canal (discharge temperature is 70 F plus the 20 degree rise in the condensers or 90 F (32.22 C). The thermal plume would provide additional exposure above the lethal threshold, minus 2 C (29.5 C or 85.1 F) of more than four hours.

$$1 \geq \frac{60}{10^{[34.3649-0.9789(32.22+2)]}}$$

$$1 \geq 8.15$$

Conclusion:

Juvenile bass would not survive to the end of the discharge canal.

Dilution

Criterion applied to entrainment in the system employing rapid dilution.

$$1 \geq \frac{1.2}{10^{[34.3649-0.9789(32.22+2.0)]}}$$

$$1 \geq \frac{1.2}{7.36}$$

Travel time in piping to discharge is assumed to be 1 min., and temperature drop to below the lethal threshold minus 2 C (29.5 C or 85.1 F) is about 10 sec. (Pritchard, 1971).[312]

Conclusion

Juvenile bass would survive this thermal exposure:

$$1 \geq 0.1630$$

By using the equation in the following form,

$$\log(\text{time}) = a + b\,(\text{temp.}+2)$$

the length of time that bass could barely survive the expected temperature rise could be calculated, thus allowing selection of an appropriate discharge system. For example:

$$\log(\text{time}) = 34.3649 - 0.9789\,(34.22)$$
$$\log(\text{time}) = 0.8669$$
$$\text{time} = 7.36$$

This would be about 1,325 feet of canal flowing at 3 ft/sec.

It is apparent that a long discharge canal, a nonrecirculating cooling pond, a very long offshore pipe, or delayed dilution in a mixing zone (such as the one promoting surface cooling) could prolong the duration of exposure of pumped organisms and thereby increase the likelihood of damage to them. Precise information on the travel times of the cooling water in the discharge system is needed to conduct this analysis.

The calculations have ignored changing temperatures in the thermal plume, because the canal alone was lethal, and cooling in the plume with rapid dilution was so rapid that the additional exposure was only for 10 seconds (assumed to be at the discharge temperature the whole time). There may be other circumstances under which the effect of decreasing exposure temperature in the plume may be of interest.

Effects of changing temperatures in the plume can be estimated by summing the effects of incremental exposures for short time periods (Fry et al. 1946[281]). For example, the surface cooling plume of Figures III–7 and III–8 could be considered to be composed of several short time spans, each with an average temperature, until the temperature had dropped to the upper lethal threshold minus 2 C for the juvenile bass. Each time period would be calculated as if it were a single exposure, and the calculated values for all time periods would be summed and compared with unity, as follows:

$$\frac{\text{time}_1}{10^{[a+b(\text{temp.}_1+2)]}}+\frac{\text{time}_2}{10^{[a+b(\text{temp.}_2+2)]}}+\cdots\frac{\text{time}_n}{10^{[a+b(\text{temp.}_n+2)]}}$$

The surface cooling plume of Figure III–6 (exclusive of the canal) could be considered to consist of 15 min at 89.7 F (32.06 C), 15 min at 89.2 F (31.78 C), 15 min at 88.7 F (31.4 C), 15 min at 88.2 F (31.22 C), 15 min at 87.8 F (31.00 C), until the lethal threshold for 70 F acclimation minus 2 C (85.1 F) was reached. The calculation would proceed as follows:

$$1 \geq \frac{15}{10^{[34.3649-0.9789(32.06+2)]}}+\frac{15}{10^{[34.3649-0.9789(31.78+2)]}}+\cdots$$

In this case, the bass would not survive through the first 15-minute period. In other such calculations, several steps would have to be summed before unity was reached (if not reached, the plume would not be detrimental).

Entrainment in the Plume

Organisms mixed with the thermal plume during dilution will also receive thermal shocks, although the maximum temperatures will generally be less than the discharge temperature. The number of organisms affected to some degree may be significantly greater than the numbers actually pumped through the plant. The route of maximum thermal exposure for each plume is indicated in Figure III–7 by a dashed line. This route should be analyzed to determine the maximum reproducible effect.

Detrimental effects of these exposures can also be judged by using the criterion for short-term exposures to extreme temperatures. The analytical steps were outlined above for estimating the effects on organisms that pass through the thermal plume portions of the entrainment thermal pattern. There would have been no mortalities of the largemouth bass from entrainment in the plume with rapid dilution, due to the short duration of exposure (about 10 seconds). Any bass that were entrained in the near-shore portions of the larger plume, and remained in it, would have died in less than 15 minutes.

Bottom Organisms Impacted by the Plume

Bottom communities of invertebrates, algae, rooted aquatic plants, and many incubating fish eggs can be exposed to warm plume water, particularly in shallow environments. In some circumstances the warming can be continuous, in others it can be intermittent due to changes in plume configuration with changes in currents, winds, or other factors. Clearly a thermal plume that stratifies and occupies only the upper part of the water column will have least effect on bottom biota.

Several approaches are useful in evaluating effects on the community. Some have predictive capability, while others are suitable largely for identifying effects after they have occurred. The criterion for short-term exposures identified relatively brief periods of detrimental high temperatures. Instead of the organism passing through zones of elevated temperatures, as in the previous examples, the organism is sedentary, and the thermal pulse passes over it. Developing fish eggs may be very sensitive to such changes. A brief pulse of high temperature that kills large numbers of organisms may affect a bottom area for time periods far longer than the immediate exposure time. Repeated sublethal exposures may also be detrimental, although the process is more complex than straight-forward summation. Analysis of single exposures proceeds exactly as described for plume entrainment.

The criterion for prolonged exposures is more generally applicable. The maximum tolerable weekly average temperature may be determined by the organisms present and the phase of their life cycle. In May, for example, the maximum heat tolerance temperature for the community may be determined by incubating fish eggs or fish fry on the bottom. In July it may be determined by the important resident invertebrate species. A well-designed thermal discharge should not require an extensive mixing zone where these criteria are exempted. Special criteria for reproductive processes may have to be applied, although thermal discharges should be located so that zones important for reproduction—migration, spawning, incubation—are not used.

Criteria for species diversity provide a useful tool for identifying effects of thermal changes after they have occurred, particularly the effects of subtle changes that are a result of community interactions rather than physiological responses by one or more major species. Further research may identify critical temperatures or sequences of temperature changes that cannot be exceeded and may thereby provide a predictive capability as well. (See Appendix II–B.)

Mixed Water Body (or major region thereof)

This is the region most commonly considered in establishing water quality standards, for it generally includes the major area of the water body. Here the results of thermal additions are observed as small temperature increases over a large area (instead of high temperatures locally at the discharge point), and all heat sources become integrated into the normal annual temperature cycle (Figure III–6 and Figure III–7 insert).

Detrimental high temperatures in this area (or parts of it) are defined by the criteria for maximum temperatures for prolonged exposure (warm and cool months) for the most sensitive species or life stage occurring there, at each time of year, and by the criteria for reproduction.

For example, in the lake with the hypothetical power station, there may be 40 principal fish species, of which half are considered important. These species have spawning temperatures ranging from 5 to 6 C for the sauger (*Stizostedion canadense*) to 26.7 C for the spotted bullhead (*Ictalurus serracanthus*). They also have a similar range of temperatures required for egg incubation, and a range of maximum temperatures for prolonged exposures of juveniles and adults. The requirements, however, may be met any time within normal time spans, such as January 1 to 24 for sauger spawning, and March 25 to April 29 for smallmouth bass spawning. Maximum temperatures for prolonged exposures may increase steadily throughout a spring period. To predict effects of thermal discharges the pertinent temperatures for reproductive activities and maximum temperatures for each life stage can be plotted over a 12-month period such as shown in Fig. III–6. A maximum annual temperature curve can become apparent when sufficient biological data are available. Mount (1970)[306] gives an example of this type of analysis.

Discharge Canal

Canals or embayments that carry nearly undiluted condenser cooling water can develop biological communities that are atypical of normal seasonal communities. Interest in these areas does not generally derive from concern for a balanced ecosystem, but rather from effects that the altered communities can have on the entire aquatic ecosystem.

The general criteria for nuisance organisms may be applicable. In the discharge canals of some existing power stations, extensive mats of temperature-tolerant blue-green algae grow and periodically break away, adding a decomposing organic matter to the nearby shorelines.

The winter criterion for maximum temperatures for prolonged exposures identifies the potential for fish kills due to rapid decreases in temperature. During cold seasons particularly, fish are attracted to warmer water of an enclosed area, such as a discharge canal. Large numbers may reside there for sufficiently long periods to become metabolically acclimated to the warm water. For any acclimation temperature there is a minimum temperature to which the species can be cooled rapidly and still survive (lower incipient lethal temperature). These numerical combinations, where data are available, are found in Appendix II–C. There would be 50 per cent mortality, for example, if largemouth bass acclimated in a discharge canal to 20 C, were cooled to 5.5 C or below. If normal winter ambient temperature is less than 5.5 C, then the winter maximum should be below 20 C, perhaps nearer 15 C. If it is difficult to maintain the lower temperatures, fish should be excluded from the area.

TOXIC SUBSTANCES

ORGANIC MERCURY

Until recently, mercury most commonly entered the aquatic environment by leaching from geological formations and by water transport to streams and lakes. Since the industrial revolution, however, increasing amounts of mercury have been added to the aquatic environment with waste products from manufacturing processes or through improper disposal of industrial and consumer products. In addition, large quantities of mercury enter the environment when ores are smelted to recover such metals as copper, lead, and zinc (Klein 1971),[343] and when fossil fuels are burned. Whereas the maximum amount of mercury released by weathering processes is approximately 230 metric tons per year worldwide, the amount released by the burning of coal is on the order of 3000 tons per year; and a further quantity, probably comparable to 3000 tons, is emitted from industrial processes (Joensuu 1971).[341]

In urban and industrial areas consumer products containing mercury are often disposed of in sewer systems. These mercury discharges, though individually small, cannot be considered insignificant, because cumulatively they add large quantities of mercury to the water courses that receive these effluents. On the average, the mercury concentration in sewage effluent is one order of magnitude greater than its concentration in the water course that receives it (D'Itri *unpublished data* 1971).[350] Based on Klein and Goldberg's 1970[344] report of mercury concentrations in samples of ocean sediments near municipal sewer out-falls, it can be calculated that in an urban area from 400 to 500 pounds of mercury per million population are discharged to receiving waters every year. The uses of mercury are varied, and its consumption is fairly large. The National Academy of Sciences (1969)[347] reported the consumption of mercury by user category.

World attention focused on the environmental mercury problem when human beings were poisoned by eating contaminated fish and shell fish during the middle and late 1950's in Minamata, Japan. Since the first occurrence of "Minamata disease" in 1953, 121 cases resulting in 46 deaths have been confirmed in the Minamata area with an additional 47 confirmed cases and 6 deaths in nearby Niigata (Takeuchi 1970).[352]

In Sweden in the 1950's, conservationists charged that the abundance of methylmercury in the environment was causing severe poisoning in seed-eating birds and their predators (Johnels et al. 1967).[342] These poisonings could be related to the use of methylmercury in seed dressings. When these seed dressings were prohibited, levels of mercury declined substantially in seed-eating animals. At about the same time, investigators found high levels of mercury in fish in waters off Sweden, practically all of it in the form of methylmercury.

Biological Methylation

Some microbes are capable of biologically synthesizing methylmercury from mercury ions (Jensen and Jernelöv 1969;[339] Wood et al. 1969;[358] Dunlap 1971;[333] Fagerström and Jernelöv 1971).[334] At low concentrations, the formation of dimethylmercury is favored in the methyl transfer reaction but at higher concentrations of mercury, the major product appears to be monomethylmercury. In any particular ecosystem, the amounts of mono- and dimethylmercury compounds are determined by the presence of microbial species, the amount of organic pollution loading, the mercury concentration, temperature, and pH (Wood et al. 1969).[358]

Biological Magnification

Aquatic organisms concentrate methylmercury in their bodies either directly from the water or through the food chain (Johnels et al. 1967;[342] Hannerz 1968;[336] Hasselrot 1968,[338] Miettinen et al. 1970[346]). Northern pike (*Esox lucius*) and rainbow trout (*Salmo gairdneri*) are able to assimilate and concentrate methylmercury directly into their muscle tissues from ingested food (Miettinen et al. 1970).[346] In general, mercury in organisms eaten by fish increases at each trophic level of the food chain (Hamilton 1971).[335] The magnitude of the bioaccumulation of mercury is determined by the species, its exposure, feeding habits, metabolic rate, age and size, quality of the water, and the degree of mercury pollution in the water. Rucker and

Amend (1969)[349] established that rainbow trout contained mercury levels of 4.0 and 17.3 μg/g in their muscle and kidney tissue after being exposed to 60 μg/l of ethylmercury for one hour a day over 10 days. Fresh water phytoplankton, macrophytes, and fish are capable of biologically magnifying mercury concentrations from water 1000 times (Chapman et al. 1968).[370] Johnels et al. (1967)[342] reported a mercury concentration factor from water to pike of 5000 or more. Johnels et al. (1967)[342] had previously shown that when mercury levels in pike muscle were below 0.2 μg/g, the level was relatively constant irrespective of weight, but above 0.2 μg/g, the concentration of mercury tended to increase with increasing age and weight.

Experiments in progress at the National Water Quality Laboratory in Duluth, Minnesota, (Mount *unpublished data* 1971)[361] indicate that when brook trout (*Salvelinus fontinalis*) are held in water containing 0.05 μg/l of methylmercury for 2 months they can accumulate more than 0.5 μg/g of mercury. This is a magnification of 10,000 times. In the same experiments, exposure to 0.03 μg/l for 5 months resulted in continuing accumulation in fish tissue with no indication of a plateau. In a group of fish held at one μg/l, some organs contained 30 μg/g. Some fresh water invertebrates have also been reported to have a 10,000 magnification (Hannerz 1968).[336]

Although the mechanisms by which mercury accumulates and concentrates have not been fully explained, at least three factors are involved: the metabolic rate of individual fish; differences in the selection of food as fish mature; and the epithelial surface of the fish (Wobeser et al. 1970;[357] Hannerz 1968).[336] The rate at which fish lose methylmercury also has considerable effect on magnification of mercury in the tissues. Miettinen et al. have shown in a series of papers (1970)[346] that the loss of methylmercury is both fast and slow in fishes. The fast loss occurs early, while mercury is being redistributed through the body, and lasts only a few weeks. The subsequent loss from established binding sites follows slowly; a half-life is estimated to be on the order of 2 years. These rates mean that fishes, and perhaps other lower vertebrates, reduce their content of methylmercury many times more slowly than do the higher terrestrial vertebrates. Man, for example, is usually considered to excrete half of any given mercury residue in about 80 days. Extremely low rates of loss have also been shown in different species of aquatic mollusks and crayfish (*Cambarus*) (Nelson 1971).[348]

Excessive mercury residues in the sediments are dissipated only slowly. Löfroth (1970)[345] estimated that aquatic habitats polluted with mercury continue to contaminate fish for as long as 10 to 100 years after pollution has stopped.

Mercury in Fresh Waters

Mercury measured in the water of selected rivers of the United States ranged from less than 0.1 μg/l to 17 μg/l. Two-thirds of the rivers contained 0.1 μg/l or less (Wallace et al. 1971).[355] The value of 0.1 μg/l is also reported as the earliest reliable estimate of mercury levels in uncontaminated fresh water (Swedish National Institute of Public Health 1971).[351] Some rivers tested by the Swedish Institute were as low as 0.05 μg/l, which was also the average mercury level in some salt waters.

Toxicity of Organic Mercury in Water

The chemical form of methylmercury administered to fish makes little difference in its toxic effect (Miettinen et al. 1970).[346] The methylmercury bound to sulfhydryl groups of proteins, as it would be in nature, is just as toxic as the free unbound ionic form.

Fish are able to survive relatively high concentrations of organomercurials for a short time with few ill effects. For example, fry of steel head trout (*Salmo gairdneri*) and fingerlings of sockeye salmon (*Oncorhynchus nerka*) are able to survive in 10 mg/l of pyridyl mercuric acetate for one hour with no toxic effects (Rucker and Whipple 1951).[350] The LC50 of pyridyl mercuric acetate for some freshwater fish ranges from 390 μg/l to 26,000 μg/l for exposures between 24 and 72 hours (Willford 1966;[356] Clemmens and Sneed 1958,[331] 1959).[332]

As the exposure times lengthen, lower concentrations of mercury are lethal. On the basis of 120-hour bioassay tests of three species of minnows, Van Horn and Balch (1955)[354] determined that the minimum lethal concentrations of pyridyl mercuric acetate, pyridyl mercuric chloride, phenyl mercuric acetate, and ethyl mercuric phosphate averaged 250 μg/l.

Recent experiments at the National Water Quality Laboratory (Mount, *personal communication* 1971)[360] indicated that 0.2 μg/l of methylmercury killed fathead minnows (*Pimephales promelas*) within 6 to 8 weeks. Toxicity data from this same laboratory on several other species including *Gammarus*, *Daphnia*, top minnow (*Fundulus* sp.) and brook trout (*Salvelinus fontinalis*) indicated that none was more sensitive than the fathead minnow.

Northern pike seem to be more sensitive. When they were reared in water containing 0.1 μg/l of methylmercury for a season and then placed in clean water, they underwent continuing mortality. Scattered mortality from this source could ordinarily not be detected in nature, because the affected fish became uncoordinated and probably would have been eaten by predators (Hannerz 1968,[336] quoted by Nelson 1971[348]).

Some species of plankton are particularly sensitive. Studies of the effect of mercury on phytoplankton species confirmed that concentrations as low as 0.1 μg/l of selected organomercurial fungicides decreased both the photosynthesis and the growth of laboratory cultures of the marine alga *Nitzschia delicatissum*, as well as of some fresh water phytoplankton species (Harriss et al. 1970).[337] Ethylmercury phosphate is lethal to marine phytoplankton at 60 μg/l, and levels as low as 0.5 μg/l drastically limit their growth (Ukeles 1962).[353] There is insufficient information about the thresholds for chronic toxicity.

Tissue Levels and Toxicity

There is almost no information on the concentrations of mercury in the tissues of aquatic organisms that are likely to cause mortality of the organisms themselves. Fish and shellfish found dead in Minamata contained 9 to 24 μg/g of mercury on the usual wet-weight basis; presumably some of these levels were lethal (Nelson 1971).[348] Miettinen et al. (1970)[346] showed that pike which had been experimentally killed by methylmercury contained from 5 to 9.1 μg/g and averaged 6.4 and 7.4 micrograms of methylmercury per gram of muscle tissue.

Discussion of Proposed Recommendations

At the present time there are not sufficient data available to determine the levels of mercury in water that are safe for aquatic organisms under chronic exposure. There have not been, for example, any experiments on the effects of chronic exposure to mercury on reproduction and growth of fish in the laboratory. Since experiments on sublethal effects are lacking, the next most useful information is on lethal effects following moderately long exposures of weeks or months. The lowest concentration shown to be lethal to fish is 0.2 μg/l of methylmercury which is lethal to fathead minnows (*Pimephales promelas*) in six weeks. Because 0.2 μg/l of methylmercury has been shown to be lethal, it is suggested that this concentration of mercury not be exceeded at any time or place in natural waters. Since phytoplankton are more sensitive, the average concentration of methylmercury in water probably should not exceed 0.05 μg/l for their protection. This recommended average is approximately equal to the supposed natural concentrations of mercury in water; hence little mercury can be added to the aquatic environment. The National Water Quality Laboratory (Mount, *unpublished data* 1971)[361] found that exposure of trout to 0.05 μg/l of methylmercury for 3 months resulted in concentrations of 0.5 μg/g, the Food and Drug Administration guideline for the maximum level for edible portions of fish flesh.

These concentrations of mercury or methylmercury in water are very low and difficult to measure or differentiate without special equipment and preparation. These low concentrations can also only be measured as total mercury. Since sediments may contain 10,000 times the amount of mercury in water, suspended solids in water can seriously affect the values found in analyses of water for mercury (Jernelöv 1972).[340] Because of these difficulties and because the real danger of mercury pollution results from a biological magnification, recommendations for mercury residues in tissues of aquatic organisms should be developed. This would make monitoring and control not only more effective and certain but also more feasible technically. Unfortunately, data are not yet available on the residue levels that are safe for the aquatic organisms themselves and for organisms higher in the food chain, such as predatory fish or fish-eating birds. It is known that concentrations of 5 to 10 μg/g are found in some fish that died of methylmercury poisoning, and that 0.01 to 0.2 μg/g is apparently a usual background level in freshwater fish. Because data are lacking for safe residue levels in aquatic food chains, it is suggested that the Food & Drug Administration guideline level of 0.5 μg/g of total mercury in edible portions of freshwater fish used as human food be the guideline to protect predators in aquatic food chains.

Hence, mercury residues should not exceed 0.5 μg/g in any aquatic organisms. If levels approaching this are found, there should be total elimination of all possible sources of mercury pollution.

No distinction has been drawn between organic and inorganic forms of mercury in these discussions because of the possibility of biological transformation to the organic phase in aquatic habitats. Since the form of mercury in water cannot be readily determined, the recommendations are primarily based upon methylmercury but expressed as total mercury.

Recommendations

Selected species of fish and predatory aquatic organisms should be protected when the following conditions are fulfilled: (1) the concentration of total mercury does not exceed a total body burden of 0.5 μg/g wet weight in any aquatic organism; (2) the total mercury concentrations in unfiltered water do not exceed 0.2 μg/l at any time or place; and (3) the average total mercury concentration in unfiltered water does not exceed 0.05 μg/l.

PHTHALATE ESTERS

The occurrence of dialkyl phthalate residues has been established in various segments of the aquatic environment of North America. Phthalate ester residues occur principally in samples of water, sediment, and aquatic organisms in industrial and heavily populated areas (Stalling 1972).[366] In fish di-*n*-butyl phthalate residues ranged from 0 to 500 μg/kg, and di-2-ethylhexyl phthalate residues were as high as 3,200 μg/kg. No well-documented information exists on the fate of phthalate compounds in aquatic environments.

Phthalate esters are widely used as plasticizers, particularly in polyvinyl chloride (PVC) plastics. The most common phthalate ester plasticizer is di-2-ethylhexyl phthalate. Di-*n*-butyl phthalate has been used as an insect repellent (Frear 1969)[362] and in pesticide formulations to retard volatilization (Schoof et al. 1963).[365] Production of dioctyl phthalate ester placticizers was estimated to be 4.10×10^8 lbs in 1970 (Neelyl 970).[363] Total phthalate ester production was reported to be 8.40×10^8 lbs in 1968, of which 4.40×10^8 lbs were dioctyl phthalate esters (Nematollahi et al. 1967).[364] Production of phthalic anhydride was estimated to be

7.60×10^8 lbs in 1970 (Neely 1970).[363] PVC plastic formulations may contain 30 to 60 parts per hundred of phthalate ester plasticizer (Nematollahi et al. 1967).[364]

Toxicity

Studies to determine the acute or chronic toxicity effects of phthalate esters or other plasticizers on aquatic organisms have only recently been undertaken (Stalling 1972).[366] For example, the acute toxicity of di-*n*-butyl phthalate to fish is extremely low compared to pesticides (Table III–14).

Daphnia magna were exposed to 0.1 μg/l of ^{14}C di-*n*-butyl phthalate and the organisms accumulated chemical residues of 600 μg/kg within 10 days, or a 6,000-fold magnification (Saunders, *unpublished data* 1971).[367] However, after transfer of the *Daphnia* to uncontaminated water, approximately 50 per cent of the di-*n*-butyl phthalate was excreted in three days. It was recently found that a concentration of 3 μg/l of di-2-ethylhexyl phthalate significantly reduced the growth and reproduction of *Daphnia magna* (Sanders *unpublished data* 1971).[367]

The acute toxicity of phthalate esters appears to be relatively insignificant, but these compounds may be detrimental to aquatic organisms at low chronic concentrations.

Recommendation

Until a more detailed evaluation is made of toxicological effects of phthalate esters on aquatic ecosystems, a safety factor of 0.1 has been applied to data for *Daphnia magna* oxicity, and a level not to exceed 0.3 μg/l should protect fish and their food supply.

POLYCHLORINATED BIPHENYLS

TABLE III-14—Acute Toxicity of Di-n-butyl Phthalate to Four Species of Fish and **Daphnia Magna.**

Species	Temperature	LC50 in μg/l 24 hr	48 hr	96 hr
Fathead minnow (Pimephales promelas)			1490	1300 (Stalling)[366]
Bluegill (Lepomis macrochirus)		1230	731	731 (Stalling)[366]
Channel catfish (Ictalurus punctatus)		3720	2910	2910 (Stalling)[366]
Rainbow trout (Salmo gairdneri)				6470 (Sanders)[367]
Daphnia magna				>5000 (Sanders)[367]

Polychlorinated biphenyls (PCB) have been found in fish and wildlife in many parts of the world and at levels that may adversely affect aquatic organisms (Jensen et al. 1969;[376] Holmes et al. 1967;[375] Koeman et al. 1969;[378] Risebrough et al. 1968).[386] The environmental occurrence, uses, and present toxicological aspects of PCB were recently reviewed by Peakall and Lincer (1970),[384] Gustafson (1970),[372] Risebrough (1970),[387] and Reynolds (1971).[385]

Biphenyls may have 1 to 10 attached chlorine atoms, making possible over 200 compounds (Gustafson 1970).[372] PCB occur as residues in fish, and presumably also in water, as mixtures of chlorinated biphenyl isomers as shown in Table III–15 (Stalling and Johnson, *unpublished data* 1970,[396] Stalling *in press*[392]).

Analysis of PCB has been accomplished by gas chromatography after separation of PCB from pesticides. A separation method has been described by Armour and Burke (1970)[389] and modified by Stalling and Huckins (1971).[391] A method using separation on a charcoal column has shown good reproducibility (Frank and Rees, *personal communication*).[395] No standardized gas-liquid chromatography method has been proposed for the analysis of mixtures of PCB in environmental samples. The solubility of these formulations in water has not been precisely determined, but it is in the range of 100 to 1,000 μg/l (Papageorge 1970).[383] Since PCB have gas chromatographic characteristics similar to many organochlorine pesticides, they can cause serious interference in the gas chromatographic determination of chlorinated insecticides (Risebrough et al. 1968).[386]

The Monsanto Company, the sole manufacturer of PCB

TABLE III-15—Composition of PCB Residues in Selected Fish Samples from the 1970 National Pesticide Residue Monitoring Program

River	Location	Species	PCB Residue as Aroclor ® type (μg/g whole body) 1232	1248	1254	1260	Total
Ohio	Cincinnati, O.	Carp Cyprinus carpio	10	75	42	6.0	133
Ohio	Cincinnati, O.	White crappie Poximus annularus	16	17	27	5.6	66
Ohio	Marietta, O.	Channel catfish Ictalurus punctatus	38	23	11	4.9	77
Ohio	Marietta, O.	Channel catfish	16	5.2	13	4.8	38
Yazoo	Redwood, Miss.	Smallmouth buffalo Ictiobus bubalus	72	....	1.4	....	73
Hudson	Poughkeepsie, N.Y.	Goldfish Carassius auratus	9	173	32	....	213
Allegheny	Natrona, Pa.	Walleye Stizostedion vitreum v.	....	5.2	25	4.6	35
Delaware	Camden, N.J.	White perch Roccus americanus	....	8.0	6.8	3.9	19
Cape Fear	Elizabeth Town, N.C.	Gizzard shad Porosoma cepedianum	19	....	2.6	1.1	23
Lake Ontario	Port Ontario, N.Y.	White perch	13	....	4.6	1.2	19
Mississippi	Memphis, Tenn.	Drum Aplodinotus grunniens	11	....	4.5	3.4	19
Merrimac	Lowell, Mass.	Drum	14	75	6.1	3.2	98

in the United States (Gustafson 1970),[372] markets eight formulations of chlorinated biphenyls under the trademarks Aroclor® 1221, 1232, 1242, 1248, 1254, 1260, 1262, and 1268. The last two digits of each formulation designate the percent chlorine. Aroclor® 1248 and 1254 are produced in greatest quantities. They are used as dielectric fluids in capacitors and in closed-system heat exchangers (Papageorge 1970).[383] Aroclor® 1242 is used as a hydraulic fluid, and Aroclor® 1260 as a plasticizer. Chlorinated terphenyls are marketed under the trademark Aroclor® 5442 and 5460, and a mixture of bi- and terphenyls is designated Aroclor® 4465. The isomer composition and chromatographic characteristics of each formulation have been described by Stalling and Huckins (1971)[391] and Bagley et al. (1970).[370] A contaminant of some PCB, especially those manufactured in Europe, are chlorinated dibenzofurans (Brungs *personal communication* 1972).[393] Although these byproducts would appear to be extremely toxic, no data are available on their toxicity to aquatic life.

Direct Lethal Toxicity

Studies of toxicity of PCB to aquatic organisms are limited. They show considerable variation of toxicity to different species, as well as variation with the chlorine content of the PCB. Nevertheless, some trends in the toxic characteristics have become apparent, principally from the work of Mayer as described below:

- The higher the per cent chlorine, the lower the apparent toxicity of PCB to fish (Mayer, *in press*).[379] This was found in 15-day intermittent-flow bioassays using bluegills (*Lepomis macrochirus*) and channel catfish (*Ictalurus punctatus*) with Aroclor® 1242, 1248, 1254. All LC50 values were in the range 10 to 300 µg/l.
- The bluegill channel catfish experiments also illustrated that all LC50 values decreased significantly when exposures continued from 15 to 20 days. The 96-hour LC50 of a PCB to fish cannot adequately measure its lethal toxicity.
- The same tests showed that the toxicity of Aroclor® 1248 doubled when the temperature was raised from 20 C to 27 C.

To invertebrates, Aroclor® 1242 has about the same acute toxicity that it has to fish. In 4- and 7-day tests (Saunders, *in press*),[389] it killed *Gammarus* at 42 µg/l and crayfish (*Cambarus*) at 30 µg/l, with values that were similar to the 15-day LC50 reported for bluegills. However, there is an extreme range in the reported short-term lethal levels of Aroclor® 1254 for invertebrates. Saunders (*in press*)[389] reported a 96-hour LC50 as 80 µg/l for crayfish and only 3 µg/l for glass shrimp (*Paleomonetes*) in 7-day tests; and Duke et al. (1970)[371] reported that as little as 0.94 µg/l killed immature pink shrimp (*Panaeus duorarum*). Part of this variation is related to exposure periods in the tests; part is no doubt the variation in species response. Again this emphasizes the point that short-term tests of acute toxicities of PCB have serious limitations.

Marine animals may be more easily killed by PCB than freshwater ones (see Section IV). When two estuarine fishes (*Lagodon rhomboides* and *Leiostomus xanthurus*) were exposed for 14 to 45 days to Aroclor® 1254, mortalities were observed at 5 µg/l (Hansen, et al. 1971).[373] This indicated a toxicity about five times greater than summarized above for freshwater fish but about the same as the toxicity for the marine crustaceans mentioned above.

Feeding Studies

Dietary exposure to PCB seems to be less of a direct hazard to fish than exposure in water. Coho salmon (*Oncorhynchus kisutch*) fed Aroclor® 1254 in varying amounts up to 14,500 µg/kg body weight per day accumulated whole body residues which were only 0.9 to 0.5 of the level in the food after 240 days of dietary exposure. Growth rates were not affected. However, all fish exposed to the highest treatment died after 240 days exposure; and thyroid activity was stimulated in all except the group treated at the lowest concentration (Mehrle and Grant *unpublished data* 1971).[394]

At present, evaluation of data from laboratory experiments indicates that exposures to PCB in water represents a greater hazard to fish than dietary exposures. However, in the environment, residue accumulation from dietary sources could be more important, because PCB have a high affinity for sediments, and therefore, they readily enter food chains (Duke et al. 1970;[371] Nimmo, et al. 1971).[382]

Residues in Tissue

It is clear that widespread pollution of major waterways has occurred, and that appreciable PCB residues exist in fish. When analyses of 40 fish from the 1970 National Pesticide Monitoring Program were made, only one of the fish was found to contain less than 1 µg/g PCB (Stalling and Mayer 1972).[390] The 10 highest residue levels in the 40 selected fish ranged from 19 µg/g to 213 µg/g whole body weight.

By contrast, residues measured in ocean fish have been generally below 1 µg/g (Risebrough 1970;[387] Jensen, et al. 1969).[376] Between the ranges in freshwater fish and those in marine fish are the levels of PCB found in seals (Jensen et al. 1969;[376] Holden 1970),[374] and in the eggs of fish-eating birds in North America (Anderson et al. 1969;[368] Mulhern et al. 1971;[380] Reynolds 1971).[385]

In laboratory experiments, crustacea exposed to varying levels of Aroclor® 1254 in the water concentrated the PCB within their bodies more than 20,000 times. The tissue residues may sometimes reach an equilibrium, and in *Gammarus fasciatus* PCB did not concentrate beyond 27,000 times despite an additional 3-week exposure to 1.6 µg/l Aroclor® (Saunders 1972).[388] In contrast, PCB residues in crayfish did not reach equilibrium after a 28-day exposure.

PCB concentration factors by two estuarine fishes, *Lagondon rhomboides* and *Leiostomus xanthurus*, were similar to that described above for crustaceans, i.e., about 10,000 to 50,000 times the exposure levels in water (Hansen et al. 1971).[373] It is important to note that these accumulations occurred at water concentrations of PCB that killed the fish in 15 to 45 days.

Also similar were the accumulation ratios of 26,000 to 56,000 for bluegills (*Lepomis macrochirus*) chronically exposed to 2 to 15 µg/l of Aroclor® 1248 and 1254. Fathead minnows (*Pimephales promelas*) chronically exposed to Aroclor® 1242 and 1254 for 8 weeks concentrated PCB 100,000 and 200,000 times the exposure levels, respectively. Residues of 50 µg/l (whole body) resulted from exposure for 8 weeks to 0.3 µg/l Aroclor® 1254 (Nebeker et al. 1972).[381] These experiments with bluegills also indicated that the maximum levels of PCB were generally related to the concentration of PCB in the water (50,000–200,000 times higher) to which they were exposed (Stalling and Huckins *unpublished data* 1971).[397]

Effects on Reproduction

PCB residues in salmon eggs are apparently related to mortality of eggs. In preliminary investigations in Sweden, Jensen and his associates (1970)[377] reported that when residues in groups of eggs ranged from 0.4 to 1.9 µg/g on a whole-weight basis (7.7 to 34 µg/g on a fat basis), related mortalities ranged from 16 per cent up to 100 per cent.

PCB concentrations in the range of 0.5 to 10 µg/l in water interfered with reproduction of several aquatic animals according to recent work of Nebeker et al. (1971).[381] About 5 µg/l of Aroclor® 1248 was the highest concentration that did not affect reproduction of *Daphnia magna* and *Gammarus pseudolimnaeus*. In tests of reproduction by fathead minnows (*Pimephales promelas*) all died when exposed chronically to greater than 8.3 µg/l of either Aroclor® 1242 or Aroclor® 1254. Reproduction occurred at and below 5.4 µg/l Aroclor® 1242, and at and below 1.8 µg/l of Aroclor® 1254.

The association between residue levels and biological effects in aquatic animals is scarcely known, but the work of Jensen et al. (1970)[377] suggested that about 0.5 µg/g of PCB in whole salmon eggs might be the threshold for egg mortality. Such a level in eggs would be associated with levels in general body tissue (e.g., muscle) of 2.5 to 5.0 µg/g. The residue in muscle corresponded to the present Food and Drug Administration level for allowable levels of PCB in fish used as human food. Residues measured in the survey by the 1970 National Pesticide Monitoring Program were generally above 5 µg/g.

Applying a minimal safety factor of 10 for protection of the affected population, and for protection of other species higher in the food chain, would yield a maximum permissible tissue concentration of 0.5 µg/g in any aquatic organism in any habitat affected by PCB.

General Considerations and Further Needs

Another means of control would be justified in view of the toxicity of PCB, the lack of knowledge about how it first enters natural ecosystems as a pollutant, and its apparent distribution in high concentrations in freshwater fish in the United States. This method would be to regulate the manufacture of PCB and maintain close control of its uses to avoid situations where PCB is lost to the environment. The Monsanto Company recently restricted the sale of PCB for uses in which disposal of the end products could not be controlled, as with plasticizers (Gustafson 1970).[372]

Basis for Recommendations

For PCB levels in water, the most sensitive reaction shown by aquatic organisms is to the lethal effects of low concentrations continually present in water for long periods (weeks or months). Concentrations in the range of 1 to 8 µg/l have been shown to be lethal to several animals.

The work of Hansen, et al. (1971)[373] and Stallings and Huckins (*unpublished data* 1971)[391] indicates that concentrations of 0.01 µg/l of PCB in water over periods of up to 36 weeks could lead to dangerous levels of PCB in the tissues of aquatic organisms. Accumulation by factors of 75,000 to 200,000 times is indicated by their work. If the higher ratio is taken, 0.01 µg/l in water might result in 2.0 µg/g in flesh on whole fish basis. This is comparable to the residue level in salmon eggs associated with complete mortality of embryos. Therefore, a concentration is recommended that is reduced by a factor of 5, or 0.002 µg/l. In addition, a control based on residue levels is required, as well as one based on PCB in the water.

Recommendations

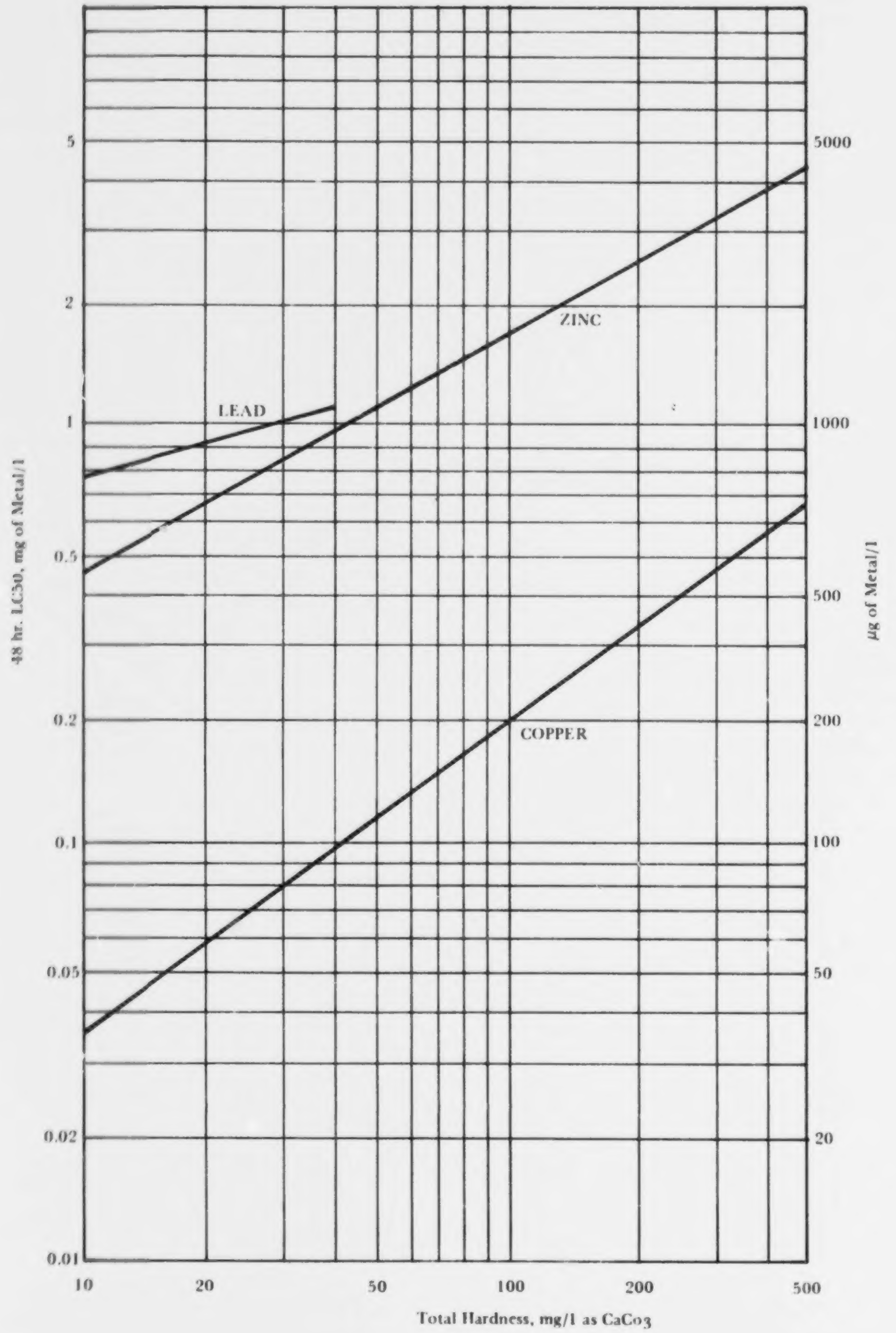
Aquatic life should be protected where the maximum concentration of total PCB in unfiltered water does not exceed 0.002 µg/l at any time or place, and the residues in the general body tissues of any aquatic organism do not exceed 0.5 µg/g.

METALS

General Data

Several reviews of the toxicity of metals are available (e.g., Skidmore 1964;[428] McKee and Wolf 1963;[415] Doudoroff and Katz 1953).[406] Some of the most relevant research is currently in progress or only recently completed. Some deals with chronic effects of metals on survival, growth, and reproduction of fish and other organisms. The completed studies have estimated safe concentrations, and from these application factors have been derived as defined in the discussion of bioassays (pp. 118–123).

The important relation between water hardness and lethal toxicity is well documented for some metals (see Figure III–9). For copper, the difference in toxicity may



Brown 1968;[401] Lloyd and Herbert 1960[414]

FIGURE III-9—The 48-Hour Lethal Concentrations of Three Heavy Metals for Rainbow Trout* (Salmo gairdneri). *(Similar Relationships Exist for Other Species of Fish.)

not be related to the difference in hardness per se, but to the difference in alkalinity of the water that accompanies change in hardness (Stiff 1971).[434] Nevertheless, the relation to hardness is a convenient and accepted one. The hardness classification developed by the U.S. Geological Survey is the following:

Soft	0– 60 mg/l (hardness as $CaCO_3$)
Moderately hard	61–120 mg/l
Hard	in excess of 120 mg/l

There are many chemical species of metals in water; some are toxic to aquatic life, others are not. Hydrogen ion concentration in water is extremely important in governing the species and solubility of metals and therefore the lethal toxicity. At high pH, many heavy metals form hydroxides or basic carbonates that are relatively insoluble and tend to precipitate. They may, however, remain suspended in the water as fine particles (O'Connor et al. 1964;[421] Stiff 1971).[434]

The toxicity of suspended hydroxides of metal depends on the particular situation. For example, suspended zinc has been found to be nontoxic (Sprague 1964a&b),[429,430] equally as toxic as dissolved zinc (Lloyd 1960)[412] and more toxic than dissolved zinc (Mount 1966).[417] This indicates that suspended zinc is at least potentially poisonous, and therefore the total metal measured in the water should be considered toxic. It is difficult to predict the effect of pH on toxicity. For example, low pH (about 5) as well as high pH (about 9) reduced toxicity of copper and zinc compared to that at neutral pH (Fisheries Research Board of Canada *unpublished data* 1971).[444] Therefore pH should be regulated in bioassays with metals in order to simulate local conditions and to explore any effect of local variation of pH.

In addition to hardness, numerous other factors influence the lethal toxicity of copper to fish. McKee and Wolf (1963)[415] and Doudoroff and Katz (1953)[406] included dissolved oxygen, temperature, turbidity, carbon dioxide, magnesium salts, and phosphates as factors affecting copper toxicity. Artificial chelating compounds such as nitrilotriacetic acid can reduce or eliminate toxic effects of zinc and other metals (Sprague 1968b)[432] and there may be natural chelating agents that would do the same thing. Certain organic ligands (Bender et al. 1970)[399] and amino acids from sewage treatment plant effluent (United Kingdom Ministry of Technology 1969)[435] also reduce the toxicity of copper by forming copper-organic complexes that do not contribute to lethal toxicity. It is safe to assume that some of these factors will influence the toxicity of other metals. In addition, the amount of metals found (at least temporarily) in living biological matter is included in most routine water analyses. At the present time, however, it is not possible to predict accurately the amount of total metal in any environment that may be lethal, biologically active, or contributory to toxicity. Consequently, the following recommendations are made.

Recommendations

Since forms or species of metals in water may change with shifts in the water quality, and since the toxicity to aquatic life may concurrently change in as yet unpredictable ways, it is recommended that water quality criteria for a given metal be based on the total amount of it in the water, regardless of the chemical state or form of the metal, except that settleable solids should be excluded from the analysis (Standard Methods 1971).[433] Additionally, hardness affects the toxicity of many metals (see Figure III-9).

Metals which have collected in the sediments can redissolve into the water, and such redissolved metals should meet the criteria for heavy metals. To protect aquatic life, amounts likely to be harmful should not occur in the sediments.

It is recommended that any metal species not specifically mentioned in this report but suspected of causing detrimental effects on aquatic life be examined as outlined in the section on Bioassays.

Aluminum

Current research by Freeman and Everhart (1971)[407] indicated that aluminum salts were slightly soluble at neutral pH; 0.05 mg/l dissolved and had no sublethal effects on fish. At pH 9, at least 5 mg/l of aluminum dissolved and this killed fingerling rainbow trout within 48 hours. However, the suspended precipitate of ionized aluminum is toxic. In most natural waters, the ionized or potentially ionizable aluminum would be in the form of anionic or neutral precipitates, and anything greater than 0.1 mg/l of this would be deleterious to growth and survival of fish.

Recommendation

Careful examination of toxicity problems should be made to protect aquatic life in situations where the presence of ionic aluminum is suspected. Aluminum may have considerably greater toxicity than has been assumed.

Cadmium

This metal is an extremely dangerous cumulative poison. In mammals (Nilsson 1970),[420] fish (Eaton *unpublished data* 1971),[442] and probably other animals, there is insidious, progressive, chronic poisoning because there is almost no excretion of the metal. In its acute lethal action on rainbow trout (*Salmo gairdneri*), Ball (1967)[398] found cadmium unusually slow. A lethal threshold of 0.01 mg/l was not discernible until seven days' exposure. Other investigators (Pickering and Gast, *in press*,[427] Eaton *unpublished data* 1971)[442] have determined lethal threshold concentrations in fathead minnows in 2 to 6 days and in bluegill in 96 hours. The chronically safe levels for both fathead minnows

(*Pimephales promelas*) (Pickering and Gast, *in press*)[427] and bluegill sunfish (*Lepomis macrochirus*) (Eaton *unpublished data* 1971)[442] in hard water (200 mg/l as $CaCO_3$) are between 0.06 and 0.03 mg/l. In these exposures, death of eggs or early larvae was one of the effects observed at the lowest unsafe concentrations tested. Recent exposures of eggs and larvae at the National Water Quality Laboratory (Duluth) in soft water (45 mg/l as $CaCO_3$) demonstrated that 0.01 mg/l was unsafe; 0.004 mg/l was safe for several warm- and coldwater fishes, including some salmonids; and the safe level for coho salmon fry (*Oncorhynchus kisutch*) was lower, i.e., between 0.004 mg/l and 0.001 mg/l (McKim and Eaton *unpublished data* 1971).[445]

Daphnia magna appeared to be very sensitive to cadmium. Concentrations of 0.0005 mg/l were found to reduce reproduction in one-generation exposures lasting three weeks (Biesinger and Christensen *unpublished data* 1971).[440] This sensitivity is probably representative of other crustaceans as well.

Recommendation

Aquatic life should be protected where levels of cadmium do not exceed 0.03 mg/l in water having total hardness above 100 mg/l as $CaCO_3$, or 0.004 mg/l in waters with a hardness of 100 mg/l or below at any time or place. Habitats should be safe for crustaceans or the eggs and larvae of salmon if the levels of cadmium do not exceed 0.003 mg/l in hard water or 0.0004 mg/l in soft water at any time or place.

Chromium

The chronic toxicity of hexavalent chromium to fish has been studied by Olson (1958),[422] and Olson and Foster (1956,[423] 1957).[424] Their data demonstrated a pronounced cumulative toxicity of chromium to rainbow trout and chinook salmon (*Oncorhynchus tshawytscha*). Duodoroff and Katz (1953)[406] found that bluegills (*Lepomis macrochirus*) tolerated a 45 mg/l level for 20 days in hard water. Cairns (1956),[403] using chromic oxide (CrO_3), found that a concentration of 104 mg/l was toxic to bluegills in 6 to 84 hours. Bioassays conducted with four species of fish gave 96-hour LC50's of hexavalent chromium that ranged from 17 to 118 mg/l, indicating little effect of hardness on toxicity (Pickering and Henderson 1966).[426]

Recently some tests of chronic effects on reproduction of fish have been carried out. The 96-hour LC50 and safe concentrations for hexavalent chromium were 33 and 1.0 mg/l for fathead minnows (*Pimephales promelas*) in hard water (Pickering *unpublished data* 1971),[446] 50 and 0.6 mg/l for brook trout (*Salvelinus fontinalis*) in soft water, and 69 and 0.3 mg/l for rainbow trout (*Salmo gairdneri*) in soft water (Benoit *unpublished data* 1971).[438] Equivalent values for trivalent chromium were little different: 27 mg/l for the 96-hour LC50, and 1.0 mg/l for a safe concentration for fathead minnows in hard water (Pickering *unpublished* data 1971).[446]

For *Daphnia* the LC50 of hexavalent chromium was reported as 0.05 mg/l, and the chronic no-effect level of trivalent chromium on reproduction was 0.33 mg/l (Biesinger and Christensen *unpublished data* 1971).[440] Some data are available concerning the toxicity of chromium to algae. The concentrations of chromium that inhibited growth for the test organisms are as follows (Hervey 1949):[410] Chlorococcales, 3.2 to 6.4 mg/l; Euglenoids, 0.32 to 1.6 mg/l; and diatoms, 0.032 to 0.32 mg/l. Patrick (*unpublished data* 1971)[447] found that 50 per cent growth reduction for two diatoms in hard and soft water occurred at 0.2 to 0.4 mg/l chromium.

Thus it is apparent that there is a great range of sensitivity to chromium among different species of organisms and in different waters. Those lethal levels reported above are 17 to 118 mg/l for fish, 0.05 mg/l for invertebrates, and 0.032 to 6.4 mg/l for algae, the highest value being 3,700 times the lowest one. The apparent "safe" concentration for fish is moderately high, but the recommended maximum concentration of 0.05 mg/l has been selected in order to protect other organisms, in particular *Daphnia* and certain diatoms which are affected at slightly below this concentration.

Recommendation

Mixed aquatic populations should be protected where the concentration of total chromium in water does not exceed 0.05 mg/l at any time or place.

Copper

Copper is known to be particularly toxic to algae and mollusks, and the implications of this should be considered for any given body of water. Based on studies of effects on these organisms, it is known that the criteria for fish protect these other forms as well. Recent work (Biesinger et al. *unpublished data* 1971)[439] indicated that the safe level of copper for reproduction and growth of *Daphnia magna* in soft water (45 mg/l as $CaCO_3$) is 0.006 mg/l, which is similar to the concentrations described below as safe for fish. The relationship of LC50 to water hardness was shown in Figure III-7 for rainbow trout (*Salmo gairdneri*).

The safe concentration of copper for reproduction by fathead minnows (*Pimephales promelas*) in hard water (200 mg/l as $CaCO_3$) was between 0.015 and 0.033 mg/l (Mount 1968),[418] and in soft water (30 mg/l as $CaCO_3$) was between 0.011 and 0.018 mg/l (Mount and Stephan 1969).[419] More recent work with fathead minnows in hard water indicated that a concentration of 0.033 mg/l would probably be safe (Brungs *unpublished data* 1971).[441] Acceptable reproduction by brook trout (*Salvelinus fontinalis*) in soft water (45 mg/l as $CaCO_3$) occurred between 0.010 and 0.018 mg/l (McKim and Benoit 1971).[416] The safe-to-lethal ratios determined in these studies varied somewhat; but that for hard water is close to 0.1 and that for soft water is approximately 0.1 to 0.2. In very soft water, typical of some northern and mountainous regions, 0.1 of the 96-hour LC50 for sensitive species would be close to what is considered a natural concentration in these waters.

Recent work indicated that avoidance reactions by fish may be as restrictive as reproductive requirements or even more so (Sprague 1964b).[430] It has been demonstrated that Atlantic salmon (*Salmo salar*) avoid a concentration of 0.004 mg/l in the laboratory.

Recommendation

Once a 96-hour LC50 has been determined using the receiving water in question and the most sensitive important species in the locality as the test organism, a concentration of copper safe to aquatic life in that water can be estimated by multiplying the 96-hour LC50 by an application factor of 0.1.

Lead

Lead has a low solubility of 0.5 mg/l in soft water and only 0.003 mg/l in hard water, although higher concentrations of suspended and colloidal lead may remain in the water. The extreme effects of water hardness on lead toxicity are demonstrated by the LC50 values in hard and soft waters. The 96-hour LC50 values in soft water (20 to 45 mg/l as $CaCO_3$) were 5 to 7 mg/l and 4 to 5 mg/l for the fathead minnow (*Pimephales promelas*) and the brook trout (*Salvelinus fontinalis*) respectively (Pickering and Henderson 1966,[426] Benoit *unpublished data* 1971).[438] Brown (1968)[401] reported a 96-hour LC50 of 1 mg/l for rainbow trout (*Salmo gairdneri*) in soft water (50 mg/l as $CaCO_3$). (See Figure III-9 for other values for this species.) The 96-hour LC50 values of lead in hard water were 482 mg/l and 442 mg/l for fathead minnow and brook trout (Pickering and Henderson 1966).[426]

There is not sufficient information on chronic toxicity of lead to fish to justify recommending values as application factors. However, preliminary information on long exposures (2 to 3 months) on rainbow trout and brook trout (Everhart *unpublished data* 1971,[443] Benoit *unpublished data* 1971)[438] indicated detrimental effects at 0.10 mg/l of lead in soft water (20 to 45 mg/l as $CaCO_3$), a safe-to-lethal ratio of less than 0.02.

Growth of guppies (*Lebistes*) was affected by 1.24 mg/l of lead (Crandall and Goodnight 1962).[405] Jones (1939)[411] and Hawksley (1967)[408] found chronic or sublethal effects on sticklebacks from lead concentrations of 0.1 and 0.3 mg/l. The conditioned behavior of goldfish (*Carassius auratus*) in a light-dark shuttlebox was adversely affected by 0.07 mg/l of lead in soft water (Weir and Hine 1970).[437]

Chronic lead toxicity was recently investigated with *Daphnia magna* (Biesinger and Christensen *unpublished data* 1971)[440] and the effect on reproduction was observed at a level of 0.03 mg/l of lead. This concentration of 0.03 mg/l, the safe level for *Daphnia*, is recommended as the criterion for protection of aquatic life. It is probably also close to the safe level for fish, because the tests described above, although somewhat preliminary, indicated that concentrations about 2 or 3 times higher had detrimental effects.

Recommendation

The concentration of lead in water should not be higher than 0.03 mg/l at any time or place in order to protect aquatic life.

Mercury

Most data about mercury involve the organic compounds (see the discussion of Organic Mercury, p. 172.) Information is available, however, for inorganic mercury in the form of mercuric ions. Short-term 96-hour bioassay studies indicated that concentrations of 1 mg/l are fatal to fish (Boetius 1960,[403] Jones 1939,[411] Weir and Hine 1970).[437] For long-term exposures of 10 days or more, mercury levels as low as 10 to 20 mg/l have been shown to be fatal to fish (Uspenskaya 1946).[436]

Recommendation

In protecting aquatic life, the recommendations for organic mercury (p. 174) also pertain here.

Nickel

The 96-hour LC50 of nickel for fathead minnows (*Pimephales promelas*) ranges from 5 mg/l in soft water (20 mg/l as $CaCO_3$) to 43 mg/l in hard water (360 mg/l as $CaCO_3$) under static test conditions (Pickering and Henderson 1966).[426] In water of 200 mg/l hardness (as $CaCO_3$), the 96-hour LC50 for fathead minnows was 26 to 31 mg/l with a chronically safe concentration between 0.8 and 0.4 mg/l (Pickering *unpublished data* 1971).[446] On the basis of this work, an application factor of 0.02 appeared to be appropriate for the protection of fish. If this factor is used, the estimated safe concentration of nickel for fathead minnows in soft water would be about 0.1 mg/l. Using static test conditions and *Daphnia magna*, Biesinger and Christensen (*unpublished data* 1971)[440] determined that a nickel concentration of 0.095 mg/l reduced reproduction during a 3-week exposure in soft water (45 mg/l as $CaCO_3$), and a nickel concentration of 0.030 mg/l had no effect. This result indicated that the sensitivity of *Daphnia magna* is comparable to that of fish.

Recommendation

Once a 96-hour LC50 has been determined using the receiving water in question and the most sensitive important species in the locality as the test organism, a concentration of nickel safe to aquatic

life in that water can be estimated by multiplying the 96-hour LC50 by an application factor of 0.02.

Zinc

The acute lethal toxicity of zinc is greatly affected by water hardness (see Figure III–7). Pickering and Henderson (1966)[426] determined the 96-hour LC50 of zinc for fathead minnows (*Pimephales promelas*) and bluegills (*Lepomis macrochirus*) using static test conditions. For fathead minnows in soft water (20 mg/l as $CaCO_3$) the LC50 was 0.87 mg/l, and in hard water (360 mg/l as $CaCO_3$) it was 33 mg/l. Bluegills were more resistant in both waters. Similarly the lethal threshold concentration was 3 or 4 times as high for coarse fish as for trout (*Salvelinus fontinalis*) (Ball 1967).[398]

The 24-hour LC50 of zinc for rainbow trout (*Salmo gairdneri*) was reduced only 20 per cent when the fish were forced to swim at 85 per cent of their maximum sustained swimming speed (Herbert and Shurben 1964).[409] The maximum effect of a reduction in dissolved oxygen from 6 to 7 mg/l to 2 mg/l on the acute toxicity of zinc was a 50 per cent increase (Lloyd 1961,[413] Cairns and Scheier 1958,[404] Pickering 1968).[425] The effects are small in comparison to the difference between acutely toxic and safe concentrations. The recommended application factor recognizes these effects.

A chronic test in hard water (200 mg/l as $CaCO_3$), involving fathead minnow reproduction, determined the safe concentration of zinc to be between 0.03 mg/l, which had no effect, and 0.18 mg/l, which caused 83 per cent reduction in fecundity (Brungs 1969).[392] Using the 96-hour LC50 of 9.2 mg/l, the ratio of the above no-effect concentration to the LC50 is 0.0034. Interpolation suggests that about 0.005 of the LC50 would cause 20 per cent reduction of fecundity, making the best estimate of a valid application factor close to 0.005.

There was a reduction in reproduction of *Daphnia magna* at a zinc concentration of 0.10 mg/l using soft water (45 mg/l as $CaCO_3$) (Biesinger and Christensen *unpublished data* 1971).[440] No effect was observed at 0.07 mg/l, which indicated that *Daphnia magna* was more resistant to zinc than the fathead minnow.

Avoidance reactions by rainbow trout in the laboratory have been caused by 0.01 of the LC50 of zinc (Sprague 1968a).[431]

Recommendation

Once a 96-hour LC50 has been determined using the receiving water in question and the most sensitive important species in the locality as the test organism, a concentration of zinc safe to aquatic life in that water can be estimated by multiplying the 96-hour LC50 by an application factor of 0.005.

PESTICIDES

Pesticides are chemicals, natural and synthetic, used to control or destroy plant and animal life considered adverse to human society. Since the 1940's a large number of synthetic organic compounds have been developed for pesticide purposes. Presently there are thousands of registered formulations incorporating nearly 900 different chemicals. Trends in production and use of pesticides indicate an annual increase of about 15 per cent, and there are predictions of increased demand during the next decade (Mrak 1969).[477] The subject of pesticides and their environmental significance has been carefully evaluated in the Report of the Secretary's Commission on Pesticides and their Relationship to Environmental Health (Mrak 1969).[477]

Methods, Rate, and Frequency of Application

Pesticides are used for a wide variety of purposes in a multitude of environmental situations. Often they are categorized according to their use or intended target (e.g., insecticide, herbicide, fungicide), but their release in the environment presents an inherent hazard to many nontarget organisms. Some degree of contamination and risk is assumed with nearly all pesticide use. The risk to aquatic ecosystems depends upon the chemical and physical properties of the pesticide, type of formulation, frequency, rate and methods of application, and the nature of the receiving system.

The pesticides of greatest concern are those that are persistent for long periods and accumulate in the environment; those that are highly toxic to man, fish, and wildlife; and those that are used in large volumes over broad areas. A list of such chemicals recommended for monitoring in the environment appears in Appendix II–F. The majority of these compounds are either insecticides or herbicides used extensively in agriculture, public health, and for household or garden purposes. In the absence of definitive data on their individual behavior and their individual effect on the environment, some generalization about pesticides is required to serve as a guideline for establishing water quality criteria to protect aquatic life. In specific instances, however, each compound must be considered individually on the basis of information about its reaction in the environment and its effect on aquatic organisms.

Sources and Distribution

The major sources of pesticides in water are runoff from treated lands, industrial discharges, and domestic sewage. Significant contributions may also occur in fallout from atmospheric drift and in precipitation (Tarrant and Tatton 1968).[485] Applications to water surfaces, intentional or otherwise, will result in rapid and extensive contamination. The persistent organochlorine pesticides have received the greatest attention in monitoring programs (Lichtenberg et al. 1970,[471] Henderson et al. 1969).[461] Their extensive distribution in aquatic systems is indicative of environmental loading from both point and nonpoint sources.

Many pesticides have a low water solubility that favors their rapid sorption on suspended or sedimented materials and their affinity to plant and animal lipids. Soluble or dispersed fractions of pesticides in the water rapidly decline after initial contamination, resulting in increased concentrations in the sediments (Yule and Tomlin 1971).[489] In streams, much of the residue is in continuous transport on suspended particulate material or in sediments (Zabik 1969).[490] The distribution within the stream flow is nonuniform because of unequal velocity and unequal distribution of suspended materials within the stream bed (Feltz et al. 1971).[454] Seasonal fluctuations in runoff and use pattern cause major changes in concentration during the year, but the continuous downstream transport tends to reduce levels in the upper reaches of streams while increasing them in the downstream areas and eventually in major receiving basins (i.e., lakes, reservoirs, or estuaries). If applications in a watershed cease entirely, residues in the stream will gradually and continuously decline (Sprague et al. 1971).[484] A similar decline would be expected in the receiving basins but at a slower rate.

In lakes the sediments apparently act as a reservoir from which the pesticide is partitioned into the water phase according to the solubility of the compound, the concentration in the sediment, and the type of sediment (Hamelink et al. 1971).[458] Dissolved natural organic materials in the water may greatly enhance the water solubility of some pesticides (Wershaw et al. 1969).[487] Some investigations indicated pesticides may be less available to the water in eutrophic systems where the higher organic content in the sediments has a greater capacity to hold pesticide residues (Lotse et al. 1968,[472] Hartung 1970[460]). This in part explained the difference in time required for some waters to "detoxify," as observed in lakes treated with toxaphene to eradicate undesirable fish species (Terriere et al. 1966).[486]

Herbicides applied to aquatic systems to control plant growths are removed from the water by absorption in the plants or sorption to the hydrosoil. The rate of disappearance from the water may be dependent upon the availability of suitable sorption sites. Frank and Comes (1967)[455] found residues of dichlobenil in soil and water up to 160 days after application. They also found that diquat and paraquat residues were persistent in hydrosoils for approximately 3 to 6 months after application. Granular herbicide treatments made on a volume basis deposit greater quantities on the hydrosoil in deep water areas than in water of less depth. The granules may supply herbicide to the water over a period of time depending upon solubility of the herbicide, concentrations in the granule, and other conditions.

Because the distribution of pesticides is nonuniform, sampling methods and frequency, as well as selection of sampling sites, must be scientifically determined (Feltz et al. 1971).[454] Pesticides found in the water in suspended particulate material and in sediments may be toxic to aquatic organisms or contribute to residue accumulation in them.

Persistence and Biological Accumulation

All organic pesticides are subject to metabolic and nonmetabolic degradation in the environment. Specific compounds vary widely in their rate of degradation, and some form degradation products that may be both persistent and toxic. Most pesticides are readily degraded to nontoxic or elementary materials within a few days to a few months; these compounds may be absorbed by aquatic organisms, but the residues do not necessarily accumulate or persist for long periods. Concentrations in the organism may be higher than ambient water levels, but they rapidly decline as water concentrations are diminished. Examples of such dynamic exchange have been demonstrated with malathion (Bender 1969),[448] methoxychlor (Burdick et al. 1968),[449] and various herbicides (Mullison 1970).[478] If degradation in water is completed within sufficient time to prevent toxic or adverse physiological effects, these nonpersistent compounds do not pose a long-term hazard to aquatic life. However, degradation rates of specific pesticides are often dependent upon environmental conditions. Considerable variation in persistence may be observed in waters of different types. Gakstatter and Weiss (1965),[456] for example, have shown that wide variations in the stability of organic phosphorous insecticides in water solutions is dependent upon the pH of the water. The half-life of malathion was reduced from about six months at pH 6 to only one to two weeks at pH 8. Repeated applications and slow degradation rates may maintain elevated environmental concentrations, but there is no indication that these compounds can be accumulated through the food chain.

Some pesticides, primarily the organochlorine compounds, are extremely stable, degrading only slowly or forming persistent degradation products. Aquatic organisms may accumulate these compounds directly by absorption from water and by eating contaminated food organisms. In waters containing very low concentrations of pesticides, fish probably obtain the greatest amount of residue from contaminated foods; but the amount retained in the tissue appears to be a function of the pesticide concentration in the water and its rate of elimination from the organism (Hamelink et al. 1971).[458] The transfer of residues from prey to predator in the food chain ultimately results in residues in the higher trophic levels many thousand times higher than ambient water levels. Examples of trophic accumulation have been described in several locations including Clear Lake, California (Hunt and Bischoff 1960),[463] and Lake Poinsett, South Dakota (Hannon et al. 1970).[459]

Residues

Samples of wild fish have often contained pesticide residues in greater concentrations than are tolerated in any commercially produced agricultural products. The highest concentrations are often found in the most highly prized fish. Coho salmon (*Oncorhyncus kisutch*) from Lake Michigan are not considered acceptable for sale in interstate commerce on the basis of an interim guideline for DDT and its metabolites set for fish by the U.S. Food and Drug Administration (Mount 1968).[476] Lake trout (*Salvelinus namaycush*) and some catches of chubs (*Coregonus kiyi* and *Coregonus hoyi*) and lake herring (*Coregonus artedi*) from Lake Michigan also exceed the guideline limits and are thus not considered acceptable for interstate commerce (Reinert 1970;[481] Michigan Department of Agriculture *personal communication*).[492] Pesticide residues in fish or fish products may enter the human food chain indirectly in other ways, as in fish oil and meal used in domestic animal feeds.

Fish may survive relatively high residue concentrations in their body fats, but residues concentrated in the eggs of mature fish may be lethal to the developing fry. Up to 100 per cent loss of lake trout (*Salvelinus namaycush*) fry occurred when residues of DDT–DDD in the eggs exceeded 4.75 mg/kg (Burdick et al. 1964).[450] A similar mortality was reported in coho salmon fry from Lake Michigan where eggs contained significant quantities of DDT, dieldrin, and polychlorinated biphenyls (Johnson and Pecor 1969;[468] Johnson 1968).[466] Johnson (1967)[467] reported that adult fish not harmed by low concentrations of endrin in water accumulated levels in the eggs that were lethal to the developing fry. Residues in fish may be directly harmful under stress conditions or at different temperature regimes. Brook trout (*Salvelinus fontinalis*) fed DDT at 3.0 mg/kg body weight per week for 26 weeks suffered 96.2 per cent mortality during a period of reduced feeding and declining water temperature. Mortality of untreated control fish during the same period was 1.2 per cent (Macek 1968).[473] Declining water temperature during the fall was believed to cause delayed mortality of salmon parr in streams contaminated with DDT (Elson 1967).[453]

In addition to the problem of pesticide residues in aquatic systems, other problems suggest themselves and remain to be investigated, including the potential of resistant fish species to accumulate levels hazardous to other species (Rosato and Ferguson 1968);[482] the potential for enhanced residue storage when fish are exposed to more than one compound (Mayer et al. 1970);[474] and the potential effect of metabolites not presently identified. The adverse effects of DDT on the reproductive performance of fish-eating birds has been well documented. (See the discussion of Wildlife, pp. 194–198.)

Levels of persistent pesticides in water that will not result in undesirable effects cannot be determined on the basis of present knowledge. Water concentrations below the practical limits of detection have resulted in unacceptable residues in fish for human consumption and have affected reproduction and survival of aquatic life. Criteria based upon residue concentrations in the tissues of selected species may offer some guidance. Tolerance levels for pesticides in wild fish have not been established, but action levels have been suggested by the U.S. Food and Drug Administration (Mount 1968).[476] However, acceptable concentrations of persistent pesticides that offer protection to aquatic life and human health are unknown.

It should also be recognized that residue criteria are probably unacceptable except on a total ecosystem basis. Residues in stream fish may meet some guidelines, but pesticides from that stream may eventually create excessive residues in fish in the downstream receiving basins. Until more is known of the effects of persistent pesticide residues, any accumulation must be considered undesirable.

Toxicity

Concentrations of pesticides that are lethal to aquatic life have often occurred in local areas where applications overlap streams or lakes, in streams receiving runoff from recently treated areas, and where misuse or spillage has occurred. Applications of pesticides to water to control noxious plants, fish, or insects have also killed desirable species. Fish populations, however, usually recovered within a few months to a year (Elson 1967).[453] The recovery of aquatic invertebrates in areas that have been heavily contaminated may require a longer period, with some species requiring several years to regain precontamination numbers (Cope 1961,[451] Ide 1967).[465] Undesirable species of insects may be the first to repopulate the area (Hynes 1961),[464] and in some instances the species composition has been completely changed (Hopkins et al. 1966).[462] Areas that are contaminated by pesticide application are subject to loss of fish populations and reduced food for fish growth (Schoenthal 1964,[483] Kerswill and Edwards 1967).[469] Where residues are persistent in bottom sediments for long periods, benthic organisms may be damaged even though water concentrations remain low (Wilson and Bond 1969).[488]

Pesticides are toxic to aquatic life over wide ranges. Great differences in susceptibility to different compounds exist between species and within species. For example, 96-hour LC50 values of 5 to 610,000 μg/l were reported for various fish species exposed to organophosphate pesticides (Pickering et al. 1962).[479] In addition to species' differences, the toxicity may be modified by differences in formulation, environmental conditions, animal size and age, and physiological condition. The effect of combinations of pesticides on aquatic organisms has not received sufficient attention. Macek (*unpublished data* 1971)[491] reported that combinations of various common pesticides were synergistic in their action on bluegill (*Lepomis macrochirus*) and rainbow trout (*Salmo gairdneri*), while others had additive effects. Several of the combinations that were found to be synergistic are recommended for insect pest control (Table III–16).

TABLE III-16—Acute Toxic Interaction of Pesticide Combinations to Rainbow Trout and Bluegills.

Pesticide combination		
Compound A	Compound B	Toxic interaction
DDT	Vapona	Additive
"	Endrin	Additive
"	Dieldrin	Additive
"	Azinphosmethyl	Additive
"	Toxaphene	Additive[a]
"	Zectran	Additive
"	BHC	Synergistic[a]
Parathion	Copper sulfate	Synergistic
"	Diazinon	Synergistic
"	DDT	Additive[a]
"	Endosulfan	Additive
"	Methoxychlor	Synergistic[a]
Malathion	Baytex	Synergistic
"	Copper sulfate	Antagonistic
"	DDT	Additive
"	EPN	Synergistic
"	Parathion	Synergistic
"	Perthane	Synergistic[a]
"	Carbaryl	Synergistic[a]
"	Toxaphene	Additive[a]
Carbaryl	Copper sulfate	Synergistic
"	DDT	Additive
"	Azinphosmethyl	Additive[a]
"	Methoxychlor	Additive
"	Parathion	Additive[a]
Methyl parathion	DDT	Additive
"	Endosulfan	Additive[a]
"	Carbaryl	Additive[a]
Bidrin	Sumithion	Additive

[a] This combination recommended for control of insect pests by the U.S. Department of Agriculture.

Note: mention of trade names does not constitute endorsement.

Most data on pesticide effects on aquatic life are limited to a few species and concentrations that are lethal in short-term tests. The few chronic tests conducted with aquatic species indicated that toxic effects occurred at much lower concentrations. Mount and Stephan (1967)[475] found the 96-hour LC50 for fathead minnows (*Pimephales promelas*) in malathion was 9,000 μg/l, but spinal deformities in adult fish occurred during a 10-month exposure to 580 μg/l. Eaton (1970)[452] found that bluegill suffered the same crippling effects after chronic exposure to 7.4 μg/l malathion and the 96-hour LC50 was 108 μg/l.

Where chronic toxicity data are available, they may be used to develop application factors to estimate safe levels. Mount and Stephan (1967)[475] have suggested using an application factor consisting of the laboratory-determined maximum concentration that has no effect on chronic exposure divided by the 96-hour LC50. Using this method, Eaton (1970)[452] showed that application factors for bluegill and fathead minnow exposed to malathion were similar despite a greater than 50-fold difference in species sensitivity. Application factors derived for one compound may be appropriate for closely related compounds that have a similar mode of action, but additional research is necessary to verify this concept. In the absence of chronic toxicity data, the application factors for many compounds must be arbitrary values set with the intention of providing some margin of safety for sensitive species, prolonged exposure, and potential effects of interaction with other compounds.

Basis for Criteria

The reported acute toxicity values and subacute effects of pesticides for aquatic life are listed in Appendix II–D. The acute toxicity values multiplied by the appropriate application factor provided the recommended criteria. The 96-hour LC50 should be multiplied by an application factor of 0.01 in most cases. The value derived from multiplying the 96-hour LC50 by a factor of 0.01 can be used as the 24-hour average concentration.

Recommended concentrations of pesticides may be below those presently detectable without additional extraction and concentration techniques. However, concentrations below those detectable by routine techniques are known to cause detrimental effects to aquatic organisms and to man. Therefore, recommendations are based on bioassay procedures and the use of an appropriate application factor.

The recommendations are based upon the most sensitive species. Permissible concentrations in water have been suggested only where several animal species have been tested. Where toxicity data are not available, acute toxicity bioassays should be conducted with locally important sensitive aquatic species, and safe levels should be estimated by using an application factor of 0.01.

Some organochlorine pesticides (i.e., DDT including DDD and DDE, aldrin, dieldrin, endrin, chlordane, heptachlor, toxaphene, lindane, endosulfan, and benzene hexachloride) are considered especially hazardous because of their persistence and accumulation in aquatic organisms. These compounds, including some of their metabolites, are directly toxic to various aquatic species at concentrations of less than one μg/l. Their accumulation in aquatic systems presents a hazard, both real and potential, to animals in the higher trophic levels, including man (Pimentel 1971,[480] Mrak 1969,[477] Kraybill 1969,[470] Gillett 1969).[457] Present knowledge is not yet sufficient to predict or estimate safe concentrations of these compounds in aquatic systems. However, residue concentrations in aquatic organisms provide a measure of environmental contamination. Therefore, specific maximum tissue concentrations have been recommended as a guideline for water quality control.

Recommendations

Organochlorine Insecticides **The recommendations for selected organochlorine insecticides are based upon levels in water and residue concentrations in whole fish on a wet weight basis. Aquatic life should be protected where the maximum con-**

centration of the organochlorine pesticide in the water does not exceed the values listed in Table III-17.

For the protection of predators, the following values are suggested for residues in whole fish (wet weight): DDT (including DDD and DDE)—1.0 mg/kg; aldrin, dieldrin, endrin, heptachlor (including heptachlor epoxide), chlordane, lindane, benzene hexachloride, toxaphene, and endosulfan—0.1 mg/kg, either singly or in combination. For further discussion, see the section on Wildlife (p. 197).

If fish and wildlife are to be protected, and where residues exceed the recommended concentrations, pesticide use should be restricted until the recommended concentrations are reached (except where a substitute pesticide will not protect human health).

Other pesticides **The recommended maximum concentrations of pesticides in freshwater are listed in Table III-18 except that where pesticides are applied to water to kill undesirable aquatic life, the values will be higher. In the latter instances, care should be taken to avoid indiscriminate use and to insure that application of the pesticide follows the prescribed methods.**

OTHER TOXICANTS

Ammonia

Ammonia is discharged from a wide variety of industrial processes and cleaning operations that use ammonia or ammonia salts. Ammonia also results from the decomposition of organic matter.

Ammonia gas is soluble in water in the form of ammonium hydroxide to the extent of 100,000 mg/l at 20 C. Ammonium hydroxide dissociates readily into ammonium

TABLE III-17—Recommended Maximum Concentrations of Organochlorine Pesticides in Whole (Unfiltered) Water, Sampled at Any Time and Any Place.[a]

Organochlorine pesticides	Recommended maximum concentration (μg/l)
Aldrin	0.01
DDT	0.002
TDE	0.006
Dieldrin	0.005
Chlordane	0.04
Endosulfan	0.003
Endrin	0.002
Heptachlor	0.01
Lindane	0.02
Methoxychlor	0.005
Toxaphene	0.01

[a] Concentrations were determined by multiplying the acute toxicity values for the more sensitive species (Appendix II-D) by an application factor of 0.01 except where an experimentally derived application factor is indicated.

TABLE III-18—Recommended Maximum Concentrations of Other Pesticides in Whole (Unfiltered) Water, Sampled at Any Time and Any Place.[a]

Organophosphate insecticides	Recommended maximum concentration (μg/l)
Abate	(b)
Azinphosmethyl	0.001
Azinphosethyl	(b)
Carbophenothion	(b)
Chlorothion	(b)
Ciodrin	0.1
Coumaphos	0.001
Demeton	(b)
Diazinon	0.009
Dichlorvos	0.001
Dioxathion	0.09
Disulfonton	0.05
Dursban	0.001
Ethion	0.02
EPN	0.06
Fenthion	0.006
Malathion	0.008
Methyl Parathion	(b)
Mevinphos	0.002
Naled	0.004
Oxydemeton methyl	0.4
Parathion	0.0004
Phorate	(b)
Phosphamidon	0.03
Ronnel	(b)
TEPP	0.4
Trichlorophon	0.002
Carbamate insecticides	**Recommended maximum concentrations (μg/l)**
Aminocarb	(b)
Bayer	(b)
Baygon	(b)
Carbaryl	0.02
Zectran	0.1
Herbicides, fungicides and defoliants	**Recommended maximum concentrations (μg/l)**
Acrolein	(b)
Aminotriazole	300.0
Balan	(b)
Bensulide	(b)
Choroxuron	(b)
CIPC	(b)
Dacthal	(b)
Dalapon	110.0
DEF	(b)
Dexon	(b)
Dicamba	200
Dichlobenil	37.0
Dichlone	0.2
Diquat	0.5
Diuron	1.6
Difolitan	(b)
Dinitrobutyl phenol	(b)
Diphenamid	(b)
2,4-D (PGBE)	(b)
2,4-D (BEE)	4.0
2,4-D (IOE)	(b)
2,4-D (Diethylamine salts)	(b)
Endothal (Disodium salt)	(b)
Endothal (Dipotassium salt)	(b)
Eptam	(b)
Fenac (Sodium salt)	45.0
Hyamine-1622	(b)
Hyamine-2389	(b)
Hydrothal-47	(b)
Hydrothal-191	(b)
Hydrothal plus	(b)
IPC	(b)
MCPA	(b)
Molinate	(b)
Monuron	(b)
Paraquat	(b)

TABLE III-18—Continued

Herbicides, fungicide and defoilants	Reccommended maximum concentration (μ/l)
Pebulate	(b)
Picloram	(b)
Propanil	(b)
Silvex (BEE)	2.5
Silvex (PGBE)	2.0
Silvex (IOE)	(b)
Silvex (Potassium salt)	(b)
Simazine	10.0
Trifluaralin	(b)
Vernolate	(b)
Botanicals	**Recommended maximum concentrations (μg/l)**
Allethrin	0.002
Pyrethrum	0.01
Rotenone	10.0

[a] Concentrations were determined by multiplying the acute toxicity values for the more sensitive species (Appendix II-D) by an application factor of 0.01 except where an experimentally derived application factor is indicated.

[b] Insufficient data to determine safe concentrations.

and hydroxyl ions as follows:

$$NH_3+H_2O\rightarrow HN_4^++OH^-$$

The equilibrium of the reaction is dependent upon pH, and within the pH range of most natural waters ammonium ions predominate (Figure III-10). Since the toxic component of ammonia solutions is the un-ionized ammonia, toxicity of ammonia solutions increases with increased pH (Ellis 1937,[497] Wuhrmann et al. 1947,[508] Wuhrmann and Woker 1948,[509] Downing and Merkens 1955[496]).

Wuhrmann (1952),[507] Downing and Merkens (1955),[496] and Merkens and Downing (1957)[505] found that a decrease in dissolved oxygen concentration increased the toxicity of un-ionized ammonia to several species of freshwater fishes. Lloyd (1961)[502] showed that the increase in toxicity of un-ionized ammonia to rainbow trout (*Salmo gairdneri*) with decreased oxygen was considerably more severe than for zinc, copper, lead, or phenol.

Much of the data on ammonia toxicity is not useable, because reporting of chemical conditions or experimental control was unsatisfactory. Ellis (1937)[497] reported that total ammonia nitrogen concentrations of 2.5 mg/l in the pH range of 7.4 to 8.5 were harmful to several fish species, but concentrations of 1.5 mg/l were not. Most streams without a source of pollution contained considerably less than 1 mg/l total ammonia. The sublethal and acutely toxic concentrations of un-ionized ammonia for various fish species are given in Table III-19.

Brockway (1950)[494] found impairment of oxygen-carrying capacity of the blood of trout at a total ammonia nitrogen concentration of 0.3 mg/l. Fromm (1970)[499] found that at total ammonia nitrogen concentrations of 5 mg/l, ammonia excretion by rainbow trout (*Salmo gairdneri*) was inhibited; at 3 mg/l the trout became hyperexcitable; and at 8 mg/l (approximately 1 mg/l un-ionized ammonia) 50 per cent were dead in 24 hours (Fromm 1970).[499] Goldfish (*Carassius auratus*) were more tolerant; at 40 mg/l of total ammonia nitrogen, 10 per cent were dead in 24 hours.

TABLE III-19—Sublethal and Acutely Toxic Concentrations of Un-Ionized Ammonia for Various Fish Species[a]

Species	Acute mortality LC50 (mg/l)	No sublethal effect (mg/l)	Author
Stickleback	1.8–2.1		Hazel et al. (1971)[500]
Striped bass (Marone saxatalis)	1.9–2.8		"
Rainbow trout	0.39	0.046	Lloyd and Orr (1969)[503]
Perch (Perca)	0.29		Ball (1967)[493]
Roach (Hesperoleucus)	0.35		"
Rudd (Scardinius)	0.36		"
Bream (Lepomis)	0.41		"
Rainbow trout	0.41		"
Rainbow trout	0.42–0.89		Lloyd and Herbert (1960)[504]
Atlantic salmon (Salmo salar)	0.38		Herbert and Shurben (1965)[501]
Rainbow trout	0.88		"
Trout		<0.27	Reichenbach-Klinke (1967)[506]
Chinook salmon (Oncorhynchus tshawytscha)		<0.006	Burrows (1964)[495]

[a] To insure a high level of protection, the mean of the 96-hour LC50's was used as a base, and an application factor of 0.05 applied to arrive at an acceptable level for most species in fresh water. Two apparently resistant species were omitted because they were far out of line with the others. After application of the factor, the resultant level is approximately half that projected from the data of Lloyd and Orr (1969).[503]

Burrows (1964)[495] found progressive gill hyperplasia in fingerling chinook salmon (*Oncorhynchus tshawytscha*) during a six-week exposure to the lowest concentration applied, 0.006 mg/l un-ionized ammonia. Reichenbach-Klinke (1967)[506] also noted gill hyperplasia, as well as pathology of the liver and blood, of various species at un-ionized ammonia concentrations of 0.27 mg/l. Exposure of carp (*Cyprinus carpio*) to sublethal un-ionized ammonia concentrations in the range of 0.11 to 0.34 mg/l resulted in extensive necrotic changes and tissue disintegration in various organs (Flis 1968).[498]

Lloyd and Orr (1969)[503] found that volume of urine production increased with exposure to increasing ammonia concentrations, but that an ammonia concentration of 12 per cent of the lethal threshold concentration resulted in no increased production of urine. This concentration of un-ionized ammonia was 0.046 mg/l for the rainbow trout used in the experiments.

Recommendation

Once a 96-hour LC50 has been determined using the receiving water in question and the most sensitive important species in the locality as the test organism, a concentration of un-ionized ammonia (NH_3) safe to aquatic life in that water can be estimated by multiplying the 96-hr LC50 by an application factor of 0.05; but no concentration greater than 0.02 mg/l is recommended at any time or place.

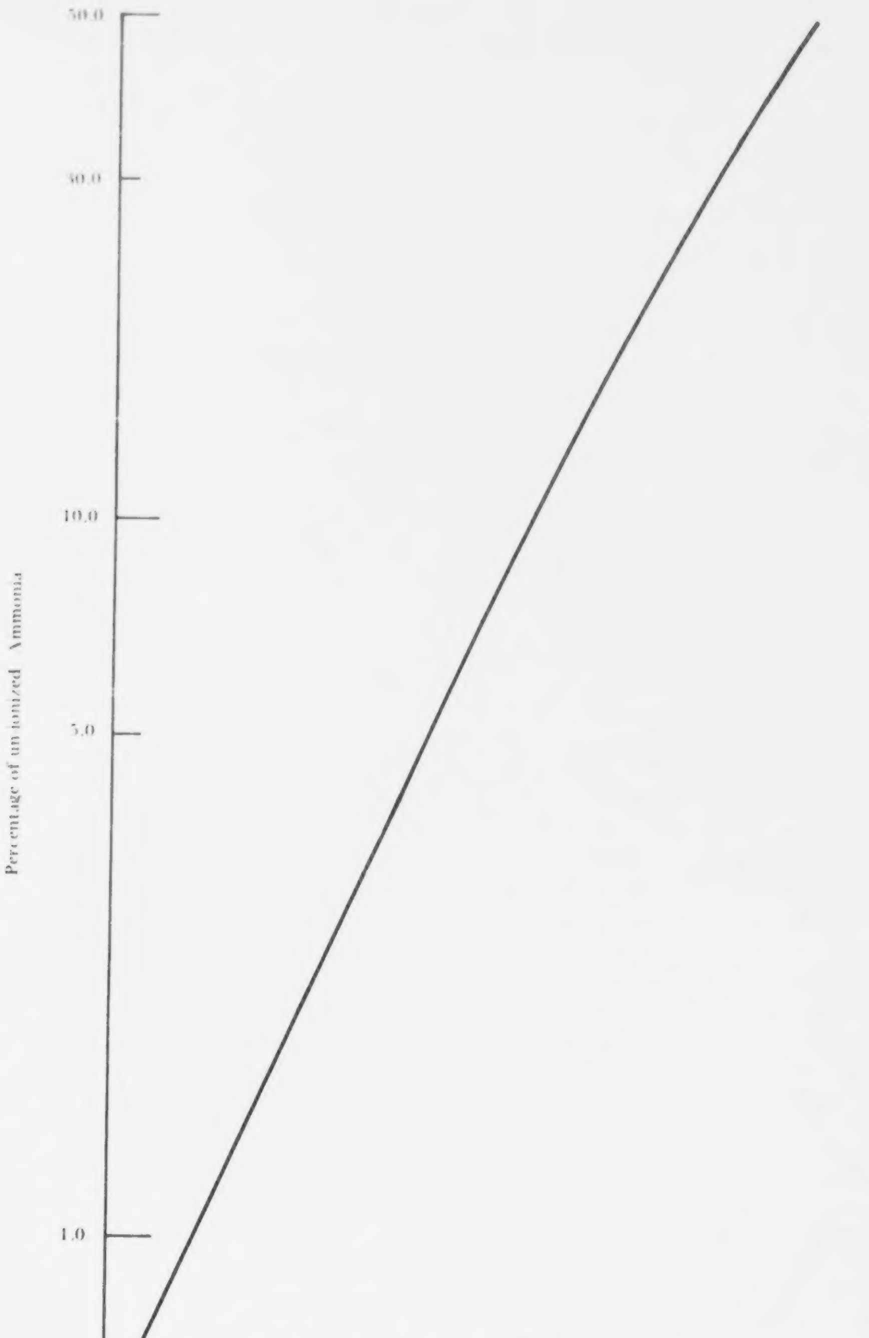


FIGURE III-10—Percentage of Un-ionized Ammonia in Ammonium Hydroxide Solutions at 20 C and Various Levels of pH

Chlorine

Chlorine and chloramines are widely used in treatment of potable water supplies and sewage-treatment-plant effluents, and in power plants, textile and paper mills, and certain other industries. Field tests conducted on caged fish in streams below a sewage outfall where chlorinated and nonchlorinated effluents were discharged showed that toxic conditions occurred for rainbow trout (*Salmo gairdneri*) 0.8 miles below the plant discharge point when chlorinated effluents were discharged (Basch et al. 1971).[511] It has also been shown that total numbers of fish and numbers of species were drastically reduced below industrial plants discharging chlorinated sewage effluents (Tsai 1968,[517] 1970).[518]

The toxicity to aquatic life of chlorine in water will depend upon the concentration of residual chlorine remaining and the relative amounts of free chlorine and chloramines. Since addition of chlorine or hypochlorites to water containing nitrogenous materials rapidly forms chloramines, problems of toxicity in most receiving waters are related to chloramine concentrations. Merkens (1958)[515] stated that toxicities of free chlorine and chloramines were best estimated from total chlorine residuals. In monitoring programs, evaluation of chlorine content of water is usually stated in terms of total chlorine residuals. Because the chlorine concentrations of concern are below the level of detection by the orthotolidine method, a more sensitive analytical technique is recommended.

The literature summarized by McKee and Wolf (1963)[514] showed a wide range of acute chlorine toxicity to various aquatic organisms, but the conditions of the tests varied so widely that estimation of generally applicable acute or safe levels cannot be derived from the combined data. It has also been demonstrated that small amounts of chlorine can greatly increase the toxicity of various industrial effluents.

Merkens (1958)[515] found that at pH 7.0, 0.008 mg/l residual chlorine killed half the test fish in seven days. The test results were obtained using the amperometric titration and the diethyl-*p*-phenylene diamine methods of chlorine analysis. Zillich (1972),[519] working with chlorinated sewage effluent, determined that threshold toxicity for fathead minnows (*Pimephales promelas*) was 0.04–0.05 mg/l residual chlorine. In two series of 96-hour LC50 tests an average of 0.05–0.19 mg/l residual chlorine was noted. Basch et al. (1971)[511] found 96-hour LC50 for rainbow trout (*Salmo gairdneri*) to be 0.23 mg/l. Arthur and Eaton (1971),[510] working with fathead minnows and *Gammarus pseudolimnaeus*, found that the 96-hour LC50 total residual chlorine (as chloramine) for *Gammarus* was 0.22 mg/l, and that all minnows were dead after 72 hours at 0.15 mg/l. After seven days exposure to 0.09 mg/l, the first fish died. The LC50 for minnows was therefore between these levels. In chronic tests extending for 15 weeks, survival of *Gammarus* was reduced at 0.04 mg/l, and reproduction was reduced at 0.0034 mg/l. Growth and survival of fathead minnows after 21 weeks was not affected by continuous exposure to 0.043 mg/l total chloramines, but fecundity of females was reduced. The highest level showing no significant effect was 0.016 mg/l. Merkens (1958)[515] postulated by extrapolation that a concentration of 0.004 mg/l residual chlorine would permit one half the test fish to survive one year. Sprague and Drury (1969)[516] have shown an avoidance response of rainbow trout to free chlorine at 0.001 mg/l.

Aquatic organisms will tolerate longer short-term exposures to much higher levels of chlorine than the concentrations which have adverse chronic effects. Brungs (1972)[512] in a review has noted that 1-hour LC50's of fish vary from 0.74 to 0.88 mg/l, and that longer short-term exposures have LC50's lower but still substantially higher than acceptable for long-term exposure. Available information, however, does not show what effect repeated exposure to these, or lower levels, will have on aquatic life.

Because *Gammarus*, an essential food for fish, is affected at 0.0034 mg/l, and a safe level is judged to be one that will not permit adverse effect on any element of the biota, the following recommendation has been made.

Recommendation

Aquatic life should be protected where the concentration of residual chlorine in the receiving system does not exceed 0.003 mg/l at any time or place. Aquatic organisms will tolerate short-term exposure to high levels of chlorine. Until more is known about the short-term effects, it is recommended that total residual chlorine should not exceed 0.05 mg/l for a period up to 30-minutes in any 24-hour period.

Cyanides

The cyanide radical is a constituent of many compounds or complex ions that may be present in industrial wastes. Cyanide-bearing wastes may derive from gas works, coke ovens, scrubbing of gases in steel plants, metal plating operations, and chemical industries. The toxicity of cyanides varies widely with pH, temperature, and dissolved oxygen concentration. The pH is especially important, since the toxicity of some cyanide complexes changes manyfold over the range commonly found in receiving waters.

"Free cyanide" (CN^- ion and HCN) occurs mostly as molecular hydrogen cyanide, the more toxic form, at pH levels of natural waters as well as in unusually acid waters. Fifty per cent ionization of the acid occurs at pH near 9.3. Free cyanide concentrations from 0.05 to 0.01 mg/l as CN have proved fatal to many sensitive fishes (Jones 1964),[527] and levels much above 0.2 mg/l are rapidly fatal for most species of fish. A level as low as 0.01 mg/l is known to have a pronounced, rapid, and lasting effect on the swimming ability of salmonid fishes.

The work of Doudoroff et al. (1966)[524] has demonstrated that the effective toxicant to fish in nearly all solutions of complex metallocyanides tested was molecular HCN, the complex ions being relatively harmless. The total cyanide content of such solutions is not a reliable index of their toxicity. The HCN derives from dissociation of the complex ions, which can be greatly influenced by pH changes. Doudoroff (1956)[523] demonstrated a more than thousandfold increase of the toxicity of the nickelocyanide complex associated with a decrease of pH from 8.0 to 6.5. A change in pH from 7.8 to 7.5 increased the toxicity more than tenfold.

Burdick and Lipschuetz (1948)[521] have shown that solutions containing the ferro and ferricyanide complexes become highly toxic to fish through photodecomposition upon exposure to sunlight. Numerous investigations have shown that toxicity of free cyanide increased at reduced oxygen concentrations (Downing 1954,[525] Wuhrmann and Woker 1955,[528] Burdick et al. 1958,[520] Cairns and Scheier 1963).[522] The toxic action is known to be accelerated markedly by increased temperature (Wuhrmann and Woker 1955,[528] Cairns and Scheier 1963),[522] but the influence of temperature during long exposure has not been demonstrated. The toxicity of the nitriles (organic cyanides) to fish varied greatly. Henderson et al. (1960)[526] found marked cumulative toxicity of acrylonitrile. Lactonitrile decomposed rapidly in water yielding free cyanide, and its high toxicity evidently was due to the HCN formed.

The toxicity of cyanide to diatoms varied little with change of temperature and was a little greater in soft water than in hard water (Patrick *unpublished data* 1971).[529] For *Nitzchia linearis*, concentrations found to cause a 50 per cent reduction in growth of the population in soft water (44 mg/l Ca-Mg as $CaCO_3$) were 0.92 mg/l (CN) at 72 F, 0.30 mg/l at 82 F, and 0.28 mg/l at 86 F. For *Navicula seminulum var. Hustedtil*, the concentrations reducing growth of the population by 50 per cent in hard water (170 mg/l Ca-Mg as $CaCO_3$) were found to be 0.36 mg/l at 72 F, 0.49 mg/l at 82 F, and 0.42 mg/l at 86 F. Cyanide appeared to be more toxic to animals than to algae.

Recommended maximum concentrations of cyanide-bearing wastes of unknown composition and properties should be determined by static and flow-through bioassays. The bioassays should be performed with dissolved oxygen, temperature, and pH held at the local water quality conditions under which cyanides are most toxic. Because the partial dissociation of some complex metallocyanide ions may be slow, static bioassays may reveal much greater toxicity than that demonstrable by the flow-through methods. On the other hand, standing test solutions of simple and some complex cyanides exposed to the atmosphere gradually lose their toxicity, because the volatile HCN escapes.

Chemical determination of the concentration of undissociated, molecular HCN alone may be the best way to evaluate the danger of free cyanide to fish in waters receiving cyanide bearing wastes. Such tests may reveal the occurrence of harmful concentrations of HCN not predictable through bioassay of the wastes. Because an acceptable concentration of HCN or fraction of a LC50 of cyanides and cyanide-bearing effluents has not yet been positively determined, a conservative estimate must be made; and because levels as low as 0.01 mg/l have proved harmful under some conditions, a factor of 0.05 should be applied to LC50 levels.

Recommendation

Once a 96-hour LC50 has been determined using the receiving water in question and the most sensitive important species in the locality as the test organism, a concentration of free cyanide (CN^-) safe to aquatic life in that water can be estimated by multiplying the 96-hour LC50 by an application factor of 0.05; but no concentration greater than 0.005 mg/l is recommended at any time or place.

Detergents

Detergents are a common component of sewage and industrial effluents derived in largest amounts from household cleaning agents. In 1965 a shift from tetrapropylene-derived alkylbenzene sulfonates (ABS) to the more biodegradable linear alkylate sulfonates (LAS) was made by the detergent industry. In current detergent formulas, LAS is the primary toxic active compound, two to four times more toxic than ABS (Pickering 1966).[534] However, toxicity of LAS disappears along with the methylene blue active substances (MBAS) response upon biodegradation (Swisher 1967).[537] Retrieval of MBAS data from the National Surveillance Stations throughout the U.S. from 1966 to the present showed that the mean of 3,608 samples was less than 0.1 mg/l. There has been a downward trend in MBAS concentrations. Only four stations reported mean concentrations greater than 0.2 mg/l.

The MBAS determination has been the routine analytical method for measurement of surfactant concentrations. Positive errors are more common than negative ones in the determination of anionic surfactants in water (Standard Methods 1971).[536] An infrared determination or a carbon absorption cleanup procedure is recommended when high MBAS concentrations are found.

Marchetti (1965)[533] critically reviewed the effects of detergents on aquatic life. Most available information on LAS toxicity relates to fish. Short term studies by a number of investigators have shown that lethal concentrations to selected fish species vary from 0.2 to 10.0 mg/l (Hokanson and Smith 1971).[532] Bardach et al. (1965)[531] reported that 10 mg/l is lethal to bullheads (*Ictalurus* sp.), and that 0.5 mg/l eroded 50 per cent of their taste buds within 24 days. Thatcher and Santner (1966)[538] found 96-hour LC50 values from 3.3 to 6.4 mg/l for five species of fish.

Pickering and Thatcher (1970)[535] found in their study of chronic toxicity that a concentration of 0.63 mg/l had no measurable effect on the life cycle of the fathead minnow (*Pimephales promelas*), while a concentration of 1.2 mg/l was lethal to the newly hatched fry. A safe level should be between 14 and 28 per cent of the 96-hour LC50. Hokanson and Smith (1971)[532] reported that a concentration of 1 mg/l was an approximate safe concentration for bluegills in Mississippi River water of good quality. Arthur (1970)[530] found that the no-effect level of LAS on *Gammarus pseudolimnaeus* was 0.2 to 0.4 mg/l. This investigator also subjected opculate and pulmonate snails to 60-week exposures of LAS and showed the toxicity levels to be 0.4 to 1.0 mg/l and greater than 4.4 mg/l, respectively.

Detergent Builders

Phosphates have been included in household detergents to increase their effectiveness, although this use has been seriously questioned recently. Nitrilotriacetate (NTA) and other builders have been tried, but most are either less effective or have been barred for reasons of potential health hazard. Available builders do not have serious direct effects on fish or aquatic organisms at concentrations likely to be encountered in receiving waters. In view of the uncertain legal status of present commercial detergents and the extensive search for adequate substitutes now in progress, recommendations for builders are not practical at this time. However, it can be stated that a satisfactory builder should be biologically degradable and nontoxic to aquatic organisms and humans, and that it should not cause aesthetic problems in the receiving water.

Recommendation

Once a 96-hour LC50 has been determined using the receiving water in question and the most sensitive important species in the locality as the test organism, a concentration of LAS safe to aquatic life in that water can be estimated by multiplying the 96-hour LC50 by an application factor of 0.05; but no concentration greater than 0.2 mg/l is recommended at any time or place.

Phenolics

Phenols and phenolic wastes are derived from petroleum, coke, and chemical industries; wood distillation; and domestic and animal wastes. Many phenolic compounds are more toxic than pure phenol: their toxicity varies with the combinations and general nature of total wastes. Acute toxicity of pure phenol varies between 0.079 mg/l in 30 minutes to minnows, and 56.0 mg/l in 96 hours to mosquito fish (*Gambusia affinis*). Mitrovic et al. (1968)[541] found a 48-hour LC50 of 7.5 mg/l to trout; they noted that exposure to 6.5 mg/l caused damage to epithelial cells in 2 hours, and extensive damage to reproductive systems in 7 days. Ellis (1937)[539] reported 1.0 mg/l safe to trout; and 0.10 mg/l was found nonlethal to bluegill (*Lepomis macrochirus*) in 48 hours (Turnbull et al. 1954).[542] These studies illustrated the wide range of phenol toxicity. There is not yet adequate documentation about chronic effects and toxicity of mixed wastes on which to base recommendations of safe levels for fish.

Phenolics affect the taste of fish at levels that do not appear to affect fish physiology adversely. Mixed wastes often have more objectionable effects than pure materials. For example, 2,4-dicholorphenol affects taste at 0.001 to 0.005 mg/l; *p*-chlorophenol at 0.01 to 0.06 mg/l; and 2-methyl, 6-chlorophenol at 0.003 mg/l. (See the discussion of Tainting Substances, p. 147.) Pure phenol did not affect taste until levels of 1 to 10 mg/l were reached (Fetterolf 1964).[540] The taste of fish in most polluted situations is adversely affected by phenolics before acute toxic effects are observed.

Recommendations

In view of the wide range of concentrations of phenolics which produce toxic effects in fish and the generally lower levels which taint fish flesh, it is recommended that taste and odor criteria be used to determine suitability of waste receiving waters to support usable fish populations. Where problems of fish kills occur or fish are subjected to occasional short-term exposure to phenolic compounds, a 96-hour LC50 should be determined using the receiving water in question and the most sensitive important fish in the locality as the test animal. Concentrations of phenolic compounds safe to fish in that water can then be estimated by multiplying the 96-hour LC50 by an application factor of 0.05; but no concentration greater than 0.1 mg/l is recommended at any time or place. Tests of other species will be necessary to protect other trophic levels.

Sulfides

Sulfides are constituents of many industrial wastes, such as those from tanneries, paper mills, chemical plants, and gas works. Hydrogen sulfide may be generated by the anaerobic decomposition of sewage and other organic matter in the water, and in sludge beds. Natural production of H_2S may also result from deposits of organic material.

When soluble sulfides are added to water, they react with hydrogen ions to form HS^- or H_2S, the proportion of each depending on the pH values. The toxicity of sulfides derives primarily from H_2S rather than the sulfide ion. The rapid combination of H_2S with other materials, including oxygen, has frequently caused investigators to overlook the importance of H_2S as it affects aquatic life, especially when it originates from sludge beds. Because water samples usually are not taken at the mud/water interface, the importance of H_2S in this habitat for fish eggs, fish fry, and

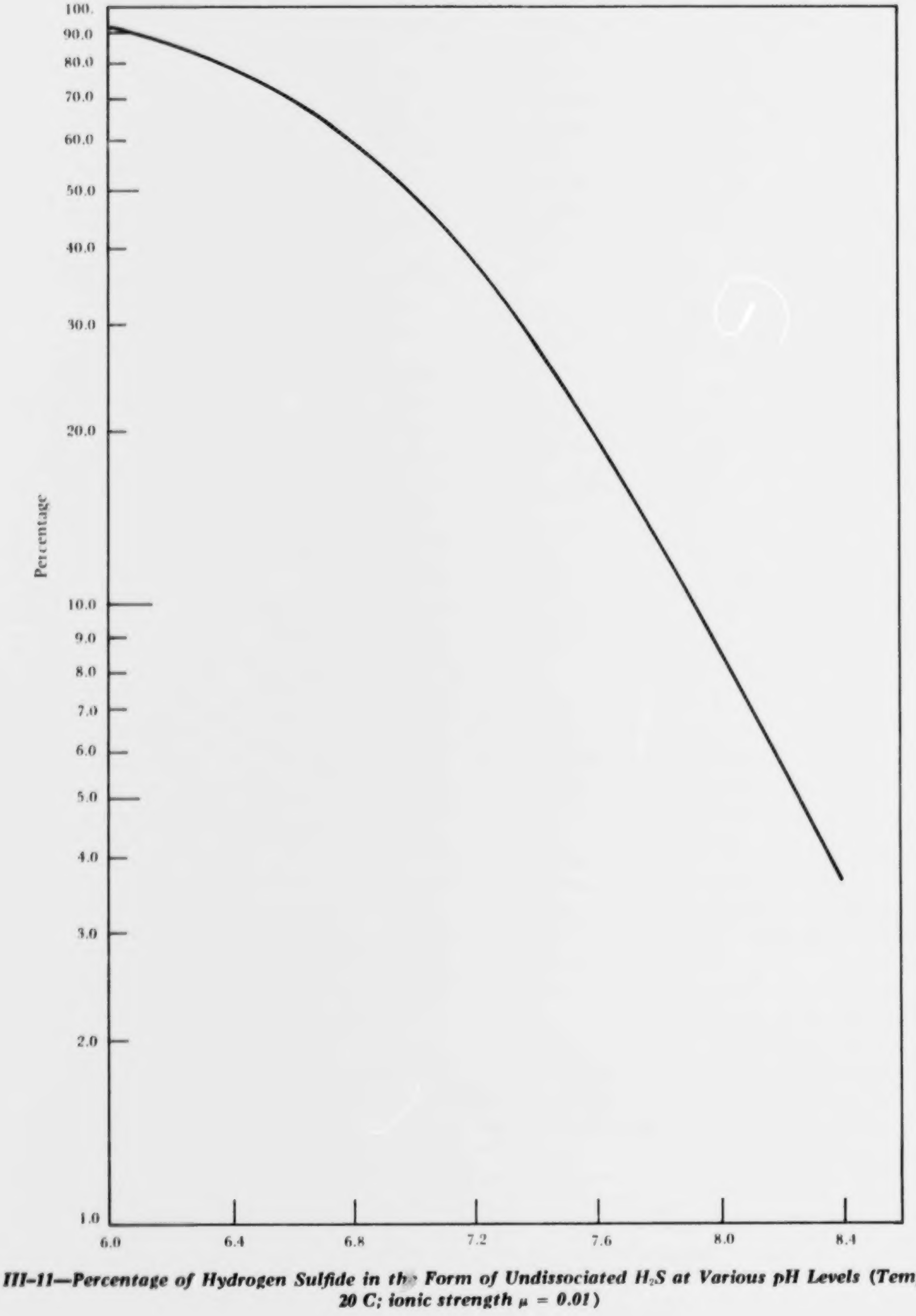


FIGURE III-11—Percentage of Hydrogen Sulfide in the Form of Undissociated H_2S at Various pH Levels (Temperature = 20 C; ionic strength μ = 0.01)

fish food organisms is often overlooked (Colby and Smith 1967).[545]

Hydrogen sulfide is a poisonous gas, soluble in water to the extent of about 4,000 mg/l at 20 C and one atmosphere of pressure (Figure III–11). Upon solution, it dissociates according to the reaction $H_2S \rightarrow HS^- + H^+$ and $HS^- \rightarrow S^{--} + H^+$. At pH 9, about 99 per cent of the sulfide is in the form of HS^-; at pH 7 it is about equally divided between HS^- and H_2S; and at pH 5 about 99 per cent is present as H_2S.

Consequently, the toxicity of sulfides increases at lower pH because a greater proportion is in the form of undissociated H_2S. Only at pH 10 and above is the sulfide ion present in appreciable amounts. In polluted situations, where the pH may be neutral or below 7.0, or where oxygen levels are low but not lethal, problems arising from sulfides or from hydrogen sulfide generated in sludge deposits will be increased.

Much available data on the toxicity of hydrogen sulfide to fish and aquatic life have been based on extremely short exposure periods and have failed to give adequate information on water quality, oxygen, and pH. Consequently, early data have suggested that concentrations between 0.3 and 4.0 mg/l permit fish to survive (Schaut 1939,[546] VanHorn 1958,[550] Bonn and Follis 1967,[544] Theede et al. 1969).[549] Recent data both in field situations and under controlled laboratory conditions demonstrated hydrogen sulfide toxicity at lower concentrations. Colby and Smith (1967)[545] found that concentrations as high as 0.7 mg/l were found within 20 mm of the bottom on sludge beds, and that levels of 0.1 to 0.02 mg/l were common within the first 20 mm of water above this layer. Walleye (*Stizostedion vitreum v.*) eggs held in trays in this zone did not hatch. Adelman and Smith (1970)[543] reported that hatching of northern pike (*Esox lucius*) eggs was substantially reduced at 0.025 mg/l of H_2S, and at 0.047 mg/l mortality was almost complete. Northern pike fry had 96-hour LC50 values that varied from 0.017 to 0.032 mg/l at normal oxygen levels (6.0 mg/l). The highest concentration of hydrogen sulfide at which no short-term effects on eggs or fry were observed was 0.814 mg/l. Smith and Oseid (*in press* 1971),[548] working on eggs, fry, and juveniles of walleyes and white suckers (*Catostomus commersonni*), and Smith (1971),[547] working on walleyes and fathead minnows (*Pimephales promelas*), found that safe levels varied from 0.0029 to 0.012 mg/l with eggs being the least sensitive and juveniles being the most sensitive in short-term tests (Table III–20). In 96-hour bioassays fathead minnows and goldfish (*Carassius auratus*) varied greatly in tolerance to hydrogen sulfide with changes in temperature. They were more tolerant at low temperatures (6 to 10 C).

TABLE III–20—96-Hour LC50 and Safe Levels Based on No Adverse Effect on Critical Life History Stages

Species		96-Hr. LC (mg/l)	Safe levels[a] (mg/l)
Northern Pike	eggs	0.037	0.014
	fry	0.026	0.004
Walleye	eggs	0.071	0.012
	fry	0.007	0.007
	juvenile	0.017	0.0037
White Sucker	eggs		0.015
	fry	0.0018	0.002
	juvenile	0.0185	0.002
Fathead minnows	juvenile	0.032 (at 20 C)	0.003
	adult	0.032	0.003
Bluegill	juvenile	0.032	0.002
	adult	0.032	0.002
Gammarus pseudolimnaeus		0.042 (10-day)	0.0033
Hexagenia limbata		0.350	

[a] Safe levels are construed to mean no demonstrable deleterious effect on survival or growth after long-term chronic exposure.

On the basis of chronic tests evaluating growth and survival, the safe level for bluegill (*Lepomis macrochirus*) juveniles and adults was 0.002 mg/l. White sucker eggs all hatched at 0.015 mg/l, but juveniles showed a negligible growth reduction at 0.002 mg/l. Safe levels for fathead minnows were between 0.002 and 0.003 mg/l. Studies on various arthropods (*Gammarus pseudolimnaeus* and *Hexagenia limbata*), useful as fish food, indicated that safe levels were between 0.002 and 0.003 mg/l (Smith 1971).[547] Some species typical of normally stressed habitats were much more resistant (*Asellus* sp.).

Recommendation

On the basis of available data, a level of undissociated hydrogen sulfide assumed to be safe for all aquatic organisms including fish is 0.002 mg/l. At a pH of 6.0 and a temperature of 13.0 C, approximately 99 per cent of the total sulfide is present as undissociated hydrogen sulfide. Therefore, to protect aquatic organisms within the acceptable limits of pH and temperature, it is recommended that the concentration of total sulfides not exceed 0.002 mg/l at any time or place.

WILDLIFE

In this report, wildlife is defined as all species of vertebrates other than fish and man. To assure the short-term and long-term survival of wildlife, the water of the aquatic ecosystem must be of the quality and quantity to furnish the necessary life support throughout the life-cycle of the species involved. In addition to the quantity, the quality of food substances produced by the aquatic environment must be adequate to support the long-term survival of the wildlife species.

Many species of wildlife require the existence of specific, complex, and relatively undisturbed ecosystems for their continued existence. Aquatic ecosystems, such as bogs, muskegs, seepages, swamps, and marshes, can exhibit marked fragility under the influence of changing water levels, various pollutants, fire, or human activity. Changes in the abundance of animal populations living in such aquatic communities can result in reactions and altered abundance of plant life, which in turn will have repercussions of other species of animal life. In general, these transitional ecosystems between land and water are characterized by very high productivity and importance for wildlife, and they should thus be maintained in that state to the greatest possible extent.

In many instances, criteria to protect fish and invertebrates or to provide water suitable for consumption by man or domestic animals will also provide the minimal requisites for some species of wildlife. This would be true for species that use water only for direct consumption or that feed on aquatic organisms to only a minor extent. For many species of wildlife, however, the setting of water quality criteria is complicated by their ecological position at the apex of complex food webs, and also by the extreme mobility of some wildlife, especially birds.

Those substances which are concentrated via food chains, such as many chlorinated hydrocarbons, present special problems for those species that occupy the apex of long food chains. In those instances, environmental levels which are safe for fish, do not necessarily convey safety to predators or even to scavengers that consume fish.

PROTECTION OF FOOD AND SHELTER FOR WILDLIFE

A number of factors can be identified that can affect specific components of the ecosystem and cause reduced food and shelter for wildlife. These factors also affect fish and other squatic life and therefore are discussed in greater detail in appropriate related subtopics.

pH

In bioassays with aquatic plants, Sincock (1968)[593] found that when the pH of the water in test vessels dropped to 4.5, reedhead-grass (*Potamogeton perfoliatus*), a valuable waterfowl food plant, died within a few days. Similarly, in Back Bay, Virginia, between August and November, 1963, the aquatic plant production declined from 164 to 13 pounds per acre. This atypical decline was immediately preceded by a decline in pH to 6.5 compared to previous midsummer readings of 7.7 to 9.2. (U.S. Bureau of Sport Fisheries and Wildlife).[601]

Recommendation

Aquatic plants of greatest value as food for waterfowl thrive best in waters with a summer pH range of 7.0 to 9.2.

ALKALINITY

Generally, waters with reasonably high bicarbonate alkalinity are more productive of valuable waterfowl food plants than are waters with low bicarbonate alkalinity. Few waters with less than 25 mg/l bicarbonate alkalinity can be classed among the better waterfowl habitats. Many waterfowl habitats productive of valuable foods, such as sago pondweed (*Potamogeton pectinatus*), widgeongrass (*Rappia maritima* and *R. occidentalis*), banana waterlily (*Castalia flava*), wild celery, (*Vallisneria americana*), and others have a bicarbonate alkalinity range of 35 to 200 mg/l.

Definitive submerged aquatic plant communities develop in waters with different concentrations of bicarbonate alkalinity. It is logical to assume that excessive and prolonged fluctuation in alkalinity would not be conducive to stabilization of any one plant community type. Sufficient experimental evidence is not available to define the effects of various degrees and rates of change in alkalinity on aquatic plant communities. Fluctuations of 50 mg/l probably would contribute to unstable plant communities. Fluctuations of this magnitude may be due to canals connecting watersheds, diversion of irrigation water, or flood diversion canals (Federal Water Pollution Control Administration 1968, hereafter referred to as FWPCA 1968).[562]

Recommendation

Waterfowl habitats should have a bicarbonate alkalinity between 30 and 130 mg/l to be productive. Fluctuations should be less than 50 mg/l from natural conditions.

SALINITY

Salinity can also affect plant communities. All saline water communities, from slightly brackish to marine, produce valuable waterfowl foods, and the most important consideration is the degree of fluctuation of salinity. The germination of seeds and the growth of seedlings are critical stages in the plant-salinity relationship; plants become more tolerant to salinity with age.

Salinities from 0.35 to 0.9 per cent NaCl in drinking water have been shown to be toxic to many members of the order Galliformes (chickens, pheasant, quail) (Krista et al. 1961,[585] Scrivner 1946,[592] Field and Evans 1946[561]).

Young ducklings were killed or retarded in growth as a result of salt poisoning by solutions equal to those found on the Suisun Marsh, California, during the summer months. Salinity maxima varied from 0.55 to 1.74 per cent, and the means varied from 0.07 to 1.26 per cent during July from 1956 to 1960 (Griffith 1962–63).[565]

Recommendation

Salinity should be kept as close to natural conditions as possible. Rapid fluctuations should be minimized.

LIGHT PENETRATION

Criteria for light penetration established in the discussions of Color (p. 130) and Settleable Solids (p. 129) should also be adequate to provide for the production of aquatic plants for freshwater wildlife.

SETTLEABLE SUBSTANCES

Accumulation of silt deposits are destructive to aquatic plants due especially to the creation of a soft, semi-liquid substratum inadequate for the anchoring of roots. Back Bay, Virginia, and Currituck Sound, North Carolina, serve as examples of the destructive nature of silt deposition. Approximately 40 square miles of bottom are covered with soft, semi-liquid silts up to 5 inches deep; these areas, constituting one-fifth of the total area, produce only 1 per cent of the total aquatic plant production (FWPCA 1968).[562]

Recommendation

Setteable substances can destroy the usefulness of aquatic bottoms to waterfowl, and for that reason, settleable substances should be minimized in areas expected to support waterfowl.

PRODUCTION OF WILDLIFE FOODS OTHER THAN PLANTS

The production of protozoans, crustaceans, aquatic insects, other invertebrates, and fish is dependent on water quality. The water quality requirements for the production of fish are dealt with elsewhere in this Section, and a normal level of productivity of invertebrates is also required for the normal production of fish that feed upon them.

While it is well known that many species of invertebrates are easily affected by low concentrations of pollutants, such as insecticides, in water (Gaufin et al. 1965,[563] Burdick et al. 1968,[555] Kennedy et al. 1970[584]), most of the field studies do not supply reliable exposure data, and most laboratory studies are of too short a duration or are performed under static conditions, allowing no reliable extrapolations to natural conditions. The general impression to be gained from these studies is that insects and crustaceans tend to be as sensitive as or more sensitive than fish to various insecticides, and that many molluscs and oligochetes tend to be less sensitive.

TEMPERATURE

The increasing discharge of warmed industrial and domestic effluents into northern streams and lakes has changed the duration and extent of normal ice cover in these northern regions. This has prompted changes in the normal overwintering pattern of some species of waterfowl. Thus, Hunt (1957)[576] details the increasing use since 1930 of the Detroit River as a wintering area for black duck (*Anas rubripes*), canvasback (*Aythya valisneria*), lesser scaup (*Aythya affines*), and redhead (*Athya americana*). In this process, waterfowl may become crowded into areas near industrial complexes with a shrinking supply of winter food. The proximity of sources of pollutants, food shortages, and low air temperatures often interact to produce unusually high waterfowl mortalities.

Recommendation

Changes in natural freezing patterns and dates should be avoided as far as possible in order to minimize abnormal concentrations of wintering waterfowl.

SPECIFIC POTENTIALLY HARMFUL SUBSTANCES

Direct Acting Substances

Oils Waterbirds and aquatic mammals, such as muskrat and otter, require water that is free from surface oil. Catastrophic losses of waterbirds have resulted from the contamination of plumages by oils. Diving birds appear to be more susceptible to oiling than other species (Hawkes 1961).[571] Heavy contamination of the plumage results in loss of buoyancy and drowning. Lower levels of contamination cause excessive heat loss resulting in an energy deficit which expresses itself in an accelerated starvation (Hartung 1967a).[567] Less than 5 mg of oil per bird can produce significant increases in heat loss. The ingestion of oils may contribute to mortalities, and this is especially true for some manufactured oils (Hartung and Hunt 1966).[569] When small quantities of oil are coated onto eggs by incubating mallards (*Anas platyrhynchos*), the likelihood of those eggs to hatch is greatly reduced (Hartung 1965).[566] Rittinghaus (1956)[591] reported an incident in which numerous Cabot's Terns (*Thalasseus sandvicensis*) and other shorebirds became contaminated with oil that had been washed on shore. Eggs which were subsequently oiled by the plumage of oiled female terns did not hatch even after 50 days of incubation. The absence of visible surface oils should protect wildlife from direct effect.

Oils can be sedimented by coating particulates on the surface and then sinking to the bottom. Sedimented oils have been associated with changes in benthic communities (Hunt 1957)[576] and have been shown to act as concentrators for chlorinated hydrocarbon pesticides (Hartung and Klingler 1970[570]).

Recommendation

To protect waterfowl, there should be no visible floating oil (see p. 146 of this Section and pp. 263–264 of Section IV).

Lead Waterfowl often mistake spent lead shot for seed or grit and ingest it. See Section IV, pp. 227–228, for a discussion of this problem.

Recommendation

The recommendation of the Marine Aquatic Life and Wildlife Panel, Section IV, (p. 228) to protect waterfowl also applies to the freshwater environment.

Botulism Poisoning Botulism is a food poisoning caused by the ingestion of the toxin of *Clostridium botulinum* of any six immunologically distinct types, designated A through F. The disease, as it occurs in epizootic proportions in wild birds, is most commonly of the C type, although outbreaks of type E botulism have been observed on the Great Lakes (Kaufman and Fay 1964[582], Fay 1966[560]).

Cl. botulinum, a widely distributed anaerobic bacterium, is capable of existing for many years in its dormant spore form, even under chemically and physically adverse environmental conditions. Its toxins are produced in the course of its metabolic activity as the vegetative form grows and reproduces in suitable media. Outbreaks occur when aquatic birds consume this preformed toxin.

The highest morbidity and mortality rates from botulism in aquatic birds have been recorded in shallow, alkaline lakes or marshes in the western United States, and outbreaks have most commonly occurred from July through September and, in some years, October. The optimum temperatures for growth of the bacterium or the toxin production, or both, have been reported as low as 25 C (Hunter et al. 1970)[577] and as high as 37 C (Quortrup and Sudheimer 1942[588]). The discrepancies are probably the result of differences in the experimental conditions under which the measurements were made and the strains of *Cl. botulinum* type C used.

The popular belief that avian botulism epizootics are associated with low water levels and consequent stagnation is not necessarily supported by facts. In three of the years of heaviest bird losses in the history of the Bear River Migratory Bird Refuge (1965, 1967, and 1971), the water supply was considerably more abundant than normal (Hunter, California Department of Fish and Game, *personal communication*; unpublished Bureau of Sport Fisheries and Wildlife reports[600]). The high water levels caused flooding of mud flats not normally under water in the summer months. Similar inundations of soil that had been dry for several years have been associated previously with outbreaks on the Bear River Refuge and in other epizootic areas. A partial explanation for these associations may be that flooding of dry ground is commonly followed by a proliferation of many species of aquatic invertebrates (McKnight 1970[587]), the carcasses of which may be utilized by *Cl. botulinum*.

Bell et al. (1955)[552] provided experimental support for an idea expressed earlier by Kalmbach (1934).[581] According to their "microenvironment concept," the bodies of invertebrate animals provide the nutrients and the anaerobic environment required by *C. botulinum* type C for growth and toxin production. These bodies would presumably also offer some protection to the bacterium and its toxin from a chemically unfavorable ambient medium. Jensen and Allen (1960)[578] presented evidence of a possible relationship between die-offs of certain invertebrate species and subsequent botulism outbreaks.

The relationship between alkalinity or salinity of the marsh and the occurrence of botulism outbreaks is not clear. Invertebrate carcasses suspended in distilled water support high levels of toxin (Bell et al. 1955).[552] Laboratory media are commonly composed of ingredients such as peptones, yeast extract, and glucose, without added salts. The medium used routinely at the Bear River Research Station for the culture of *Cl. botulinum* type C has a pH of 6.8 to 7.0 after heat sterilization. McKee et al. (1958)[586] showed that when pH was automatically maintained at a particular level in laboratory cultures of *Cl. botulinum* type C throughout the growth period, the largest amount of toxin was produced at pH 5.7, the lowest level tested. Decomposing carcasses of birds dead of botulism commonly contain very high concentrations of type C toxin, and in these cases production is ordinarily independent of the chemical composition of the marsh.

Kalmbach (1934)[581] tabulated the salt concentrations of water samples collected from 10 known botulism epizootic areas. The values ranged from 261 to 102,658 ppm (omitting the highest, which was taken from a lake where the bird losses were possibly from a cause other than botulism).

Christiansen and Low (1970)[556] recorded conductance measurements on water in the management units of the Bear River Migratory Bird Refuge and the Farmington Bay Waterfowl Management Area, both sites of botulism outbreaks varying in severity from year to year. The average conductance of water flowing into the five units of the Bear River Refuge in five summers (1959–1963) ranged from 3.7 to 4.9 millimhos per centimeter at 25 C. The readings on outflowing water from the five units ranged from 4.4 to 8.3 mmhos. Comparable figures for the three Farmington Bay units were 1.8 to 3.2 (inflow) and 3.2 to 4.8 mmhos (outflow). Thus the salinity range of the inflowing water at Bear River was comparable to that of the outflowing water at Farmington.

These data suggest that salt concentration of the water in an epizootic area is not one of the critical factors influencing the occurrence of outbreaks. If high salinity does favor their occurrence, it is probably not because of its effect on *Cl. botulinum* itself. Other possible explanations for the higher incidence of botulism in shallow, alkaline marshes are:

- Saline waters may support higher invertebrate population levels than do relatively fresh waters. (Comparisons, as they relate to avian botulism, have not been made.)
- High salinity may inhibit some of the microorganisms that compete with *Cl. botulinum* for nutrients or those that cause deterioration of the toxin.
- Salinity may have no significant effect on the invertebrates or the bacteria, but it increases the susceptibility of the birds. Cooch (1964)[557] has shown that type C botulinum toxin decreases the activity of the salt gland in ducks, reducing its capacity to eliminate salt. Birds so affected succumb to smaller doses of toxin than do those provided with fresh water.
- Outbreaks of botulism poisoning tend to be associated with or affected by insect die-offs, water temperatures above 70 F, fluctuations in water levels and elevated concentrations of dissolved solids.

Recommendation

Outbreaks of botulism poisoning tend to be associated with, or affected by insect die-offs, water temperature above 70 F, fluctuating water levels, and elevated concentrations of dissolved solids. Management of these factors may reduce outbreaks of botulism poisoning.

Substances Acting After Magnification in Food Chains

Chlorinated Hydrocarbon Pesticides

DDT and Derivatives DDT and its abundant derivatives DDE and TDE have high lipid solubility and low water solubility, and thus tend to concentrate in the lipid, i.e., living fraction of the aquatic environment (Hartung 1967b).[568] DDE is the most stable of the DDT compounds and has been especially implicated in producing thinning of egg shells, increased breakage of eggs, reproductive failure in species occupying the apex of aquatic food chains in areas with long histories of DDT usage.

Reproductive failures and local extirpation associated with egg shell thinning have been reported for several North American bird species. The phenomenon was first described and is most wide-spread for the peregrine falcon (*Falco peregrinus*) (Hickey and Anderson 1968).[574] Since then similar phenomena have been described in Brown Pelicans (*Pelecanus occidentalis*) (Anderson and Hickey 1970)[551] and species of several other families of predatory birds. Further increases of DDE in large receiving basins, such as the Great Lakes, would be expected to increase the extent of reproductive failure among predatory aquatic bird populations. Concentrations as low as 2.8 ppm *p,p′*DDE on a wet-weight basis produced experimental thinning of egg shells in the American Kestrel (*Falco sparvarius*) (Wiemeyer and Porter 1970).[599] Heath et al. (1969)[572] induced significant levels of eggshell thinning in mallards after feeding them similarly low levels of DDE. Concentrations of DDT compounds in the water of Lake Michigan have been estimated to be 1 to 3 parts per trillion (Reinert 1970)[589] (Table III-21). Concentrations that would permit the assured survival of sensitive predatory bird species are evidently much lower than that. Because such low concentrations cannot be reliably measured by present technologies and because the concentrating factor for the food chains appears to be variable or is not known, or both, a biological monitoring system should be chosen. If it is desired to protect a number of fish-eating and raptorial birds, it is essential to reduce the levels of DDE contamination, especially in large receiving basins (see Section IV).

The available data indicate that there should not be concentrations greater than 1 mg/kg of total DDT in any aquatic plants or animals in order to protect most species of aquatic wildlife. Present unpublished data indicate effects for even lower levels of DDE to some species of predatory birds (Stickel *unpublished data*).[601]

Present environmental levels vastly exceed the recommended levels in many locations, and continued direct or

TABLE III-21—Relationship of DDT and Metabolites to Eggshell Thinning

Species	Dosage* wet-weight basis	Pesticide level in eggs	Thinning Percent	Reference
Mallard	1000 mg kg single dose	N.D.†	25	Tucker & Haegele, 1970[595]
Prairie falcon (Falco mexicanus)	N.D.†	0–10 ppm DDE	ca. 5	Enderson & Berger, 1970[559]
		10–20 ppm DDE	ca. 13	
		20–30 ppm DDE	ca. 18	
		30 ppm DDE	ca. 25	
Japanese quail (Coturnix)	100 ppm o, pDDT	23.6 ppm o, pDDT 0.52 ppm DDE	4	Bitman et al., 1969[552]
	100 ppm p, p'DDT	48.0 ppm p, p'DDE	6	
Herring gull (Larus argentatus)	ca. 3.3 ppm total DDT	227 ppm total DDT	N.D.†	Keith, 1966[583]
American kestrel (Falco sparvarius)	2.8 ppm p, p'DDE	32 4 ppm DDE	10	Wiemeyer & Porter, 1970[599]
Mallard	**2.8 ppm DDE	N.D.†	11	Heath et al., 1969[572]
	**11.2 ppm DDE	N.D.†	14	

* All tests except the first one are chronic, spanning at least several months.

** Converted from dry-basis.

† Not determined.

indirect inputs of DDT would make these recommendations unattainable.

Recommendation

In order to protect most species of aquatic wildlife, the total DDT concentration on a wet-weight basis should be less than 1 mg/kg in any aquatic plants or animals. (Also see Recommendations for Pesticides, p. 185–186.)

Polychlorinated Biphenyls (PCB) Polychlorinated biphenyls are chlorinated hydrocarbons which are highly resistant to chemical or biological degradation. They have been widespread environmental contaminants (Jensen et al. 1969,[580] Risebrough et al. 1968[590]). Their biological effects at present environmental concentrations are not known. PCB's can elevate microsomal enzyme activity (Risebrough et al. 1968,[590] Street et al. 1968[594]), but the environmental significance of that finding is not clear. The toxicity of PCB is influenced by the presence of small amounts of contaminated chlorinated dibenzofurans (Vos and Koeman 1970,[596] Vos et al. 1970[597]) which are highly toxic to developing embryos.

Recommendation

Because of the persistence of PCB and their susceptibility to biological magnification, it is recommended that the body burdens of PCB in birds and mammals not be permitted to increase and that monitoring programs be instituted (see Section IV).

Mercury

Westoo (1966)[598] reported that almost all of the mercury found in fish is methyl mercury. Jensen and Jernelöv (1969)[579] showed that natural sediments can methylate ionic mercury. Mercury levels in fish in Lake St. Clair ranged between 0.4 and 3 ppm, averaging near 1.5 ppm (Greig and Seagram 1970).[564] Residues in fish-eating birds from Lake St. Clair ranged up to 7.5 ppm in a tern, and up to 23 ppm in a great blue heron (Dustman et al. 1970).[558] These residues are comparable to those found in Swedish birds that died after experimental dosing with methylmercury, and in birds that died with signs of mercury poisoning under field conditions in Scandinavian countres (Henriksson et al. 1966,[573] Borg et al. 1969,[554] Holt 1969[575]). To date, no bird mortalities due to mercury contamination have been demonstrated in the Lake St. Clair area, but body burdens of fish-eating birds are obviously close to demonstrated toxic levels. It is therefore concluded that the mercury levels in fish flesh should be kept below 0.5 ppm to assure the long-term survival of fish-eating birds. Since this level incorporates little or no safety margin for fish-eating wildlife, it is suggested that the safety of a 0.5 ppm level be reevaluated as soon as possible.

Recommendation

Fish-eating birds should be protected if mercury levels in fish do not exceed 0.5 μg/g.

Since the recommendation of 0.5 μg/g mercury in fish provides little or no safety margin for fish-eating wildlife, it is recommended that the safety of the 0.5 μg/g level be reevaluated as soon as possible.

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INTRODUCTION

The Panel on Marine Aquatic Life and Wildlife took as its prime responsibility the development of recommendations that would reasonably assure protection of the marine ecosystem. The recommendations have been discussed at various meetings of the members of the Panel and represent a consensus on the best statement that can be made in the light of present knowledge. The recommendations are not inflexible and may be modified as our understanding of the marine ecosystem improves.

Many parts of the marine ecosystem do not meet the quality requirements recommended here. As a result of man's activities, the marine ecosystem has been greatly modified; many species are excluded from areas where they were once abundant, and many areas have been closed for the harvesting of marine products as human food because of pollution. The decision as to what part, and how much, of the marine ecosystem should be protected for normal aquatic life and wildlife has political, social, and economic aspects, and such decisions cannot be based upon scientific evidence alone. Although some marine pollution problems are local in character, many are global and only the broadest possible approach can solve these problems. Food from the sea is already an important source of animal protein for human nutrition, and this continuing supply must not be diminished by pollution.

At the same time, the Panel recognizes that additions of pollutants to the oceans as by-products of our present mode of living will continue. But if pollution is kept within the boundaries and constraints which are defined in the recommendations, the Panel believes that the marine ecosystem can be protected.

In many ways the marine ecosystem is similar to the freshwater, but there are significant differences which should be briefly described. For more details which summarize the extensive literature on this subject, the reader is referred to *The Oceans* by Sverdrup, et al. (1942)[6],* *The Sea*, particularly volume 2 edited by M. N. Hill (1964),[3] and *Estuaries*, edited by G. H. Lauff (1967).[4]

The marine environment is a significant source of animal protein with an annual production of about 60 million tons fresh weight of fisheries products (Food and Agriculture Organization 1967).[2] Various estimates of the potential expansion of this harvest have been made and are summarized by Ryther (1969)[5] who concludes that the potential harvest might double this figure. Some of the existing stocks are already fished to capacity or overfished, but aquaculture (pp. 222–224) may increase world marine production.

The importance of this supply of animal protein to the world population has been emphasized by Borgstrom (1961).[1] He estimates that more than two billion people of the world's population receive 50 per cent or more of their animal protein from marine products. In the United States fish contributes only about 5 per cent of our animal protein consumption, but even so it has been estimated by Pruter (*unpublished* 1972)[8] that over ten billion pounds of commercial fish and shellfish were harvested from the estuaries and continental shelf of the United States in 1970. Furthermore, in the United States a great deal of fishmeal is used to fortify animal feeds, particularly for chickens. It is obvious that this valuable food resource of the marine environment must be sustained.

The estuaries are regions where the impact of man's activity is greatest, and they are also areas of great value for marine fish production. They serve not only as nursery areas and breeding grounds for many species of fish, but also as the regular home for the entire life cycle of some valuable species, such as oysters and crabs. Sykes (1968)[7] has estimated that 90 per cent or more of the commercial catch of finfish in some geographical regions of the United States consists of estuarine-dependent species. The estuaries are the most variable regions of the marine ecosystem (see pp. 219–221) and organisms which inhabit them are exposed to extreme variations. Since these organisms survive, they are obviously adapted to the stress imposed by these variations. During the tidal cycle, a sessile organism will be exposed to variations in temperature and salinity as the tide ebbs and flows. On a seasonal basis, because of variations in river flow, organisms at a fixed location may be exposed to fresh water during flood periods or to nearly undiluted sea water during droughts. The oscillatory nature of the tidal currents can also lead to an accumulation of pollutants within an estuary, as is discussed in the section on waste capacity of receiving waters (pp. 228–232).

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* Citations are listed at the end of the Section. They can be located alphabetically within subtopics or by their superior numbers which run consecutively across subtopics for the entire Section.

Migratory fishes must also pass through estuaries in order to reach their breeding grounds. Anadromous fishes, such as the alewife, salmon, shad, and striped bass, move upstream to breed in the highly diluted seawater or in fresh water. In contrast the catadromous species such as the eel spend their adult stages in fresh water and migrate downstream in order to breed in the open sea. Conditions within the estuaries should be maintained so that these seasonal breeding migrations are not interfered with.

The conditions in the coastal waters are less variable than those in the estuaries, but in temperate regions, the seasonal range of conditions can be considerable. The coastal waters, particularly in areas of upwelling, are the most highly productive parts of the marine environment and have been estimated by Ryther (1969)[5] to produce half of the potential marine fish production, even though they constitute only 0.1 per cent of the total area of the oceans. The coastal zones, including near shore areas of high production such as fishing banks, constitute 9.9 per cent of the area of the ocean and contribute nearly half of the world fish production. In tropical waters, the seasonal variation in conditions is less extreme than in temperate waters. However, as will be discussed in the section on temperature (p. 238) many tropical species are living near their upper extreme temperature during the summer, and this fact presents considerable problems in the disposal of waste heat in tropical areas.

The open ocean constitutes 90 per cent of the area of the world ocean and is the least variable of the marine environments. The deep sea produces only a minor fraction of the world's fish production, and this consists mainly of the large pelagic carnivores such as the tuna (Ryther 1969).[5] During the 19th century, the whale harvest was substantially greater than it is at present, but the whales captured were not as effectively utilized as they are in modern whaling methods. Many species of whales were grossly over-fished, and there is considerable question today whether some of these species can recover their original population sizes even in those cases where a complete moratorium on their capture is in effect.

The waters of the deep sea below the permanent thermocline (the depths below which seasonal temperature changes do not occur) constitute the largest and most constant environment on earth. During the history of modern oceanography, which covers the last century, no significant changes in either salinity or temperature of the deep sea have been observed, the organisms living in this abyssal environment having evolved under conditions which were presumably constant for millenia. To protect the coastal environment many proposals have been made to dump materials, such as solid waste, sewage sludge, and contaminated drege spoils in the deep sea. Since the organisms inhabiting the depths of the ocean have been exposed to a constant environment, they are not accustomed to unusual stresses which might be created by such dumping operations (see pp. 278–283). Consequently, dumping of organic wastes in the deep sea is not recommended (pp. 277, 282–283).

Development of Recommendations

In most cases, recommendations are not applicable to every local situation. The marine environment varies widely, and only an understanding of local conditions will make it possible to determine what can or cannot be added in each situation. Many materials are accumulated by marine organisms, and the concentration is often increased at higher levels of the food web. With substances that are toxic and persistent, it is the concentration in the highest predators, fish or birds, that is critical. One example is DDT and its derivatives which have accumulated in birds to levels that interfere with their breeding. Materials that decompose or are otherwise removed from the marine environment present lesser hazards.

The application of any recommendation to a local situation is unique because it requires (a) an understanding of the circulation of the water and the resultant mixing and dilution of the pollutant, (b) a knowledge of the local biological species in the environment and the identification of those that are most sensitive to the pollutant being considered, and (c) an evaluation of the transport of the material through the food web because of the possibility that the pollutant may reach concentrations hazardous either to the normal aquatic species present, or to man through his use of aquatic species as food.

The normal cycle of variation in the environment of many substances or conditions that occur naturally, such as oxygen, temperature, and nutrients, must be determined before decisions can be made as to possible permissible changes. In many estuaries and coastal waters "normal" conditions have been modified by man's activities and may already have changed to the extent that some species that might have been found at earlier times have been eliminated. In some circumstances, a recommendation may not be applicable because it may be necessary to specify no additional change beyond that which has already occurred. There is no generally applicable formula for recommendations to protect marine aquatic life and wildlife; a study of local environmental conditions is essential prior to application of the recommendations.

The Panel recognizes that what can or should be done in a given situation cannot wait for the completion of time-consuming studies. The degree of protection desired for a given location involves social and political decisions. The ecological nature and quality of each water mass proposed for modification must be assessed prior to any decision to modify. This requires appropriate information on the physical and chemical characteristics, on the distribution and abundance of species, and on the normal variations in these attributes over the annual cycle. In addition, there must be sufficient knowledge to permit useful prediction of the significant effects of the proposed pollutant on the stages in the life cycle of important species, on populations, and on the biological communities present. The possible impact of that pollutant upon the ecosystem can then be assessed. These subjects are covered in greater detail in other parts of this Section, but they are mentioned here to emphasize their primary importance in determining how the recommendations should be used in local situations.

USES OF THE MARINE SYSTEM TO BE PROTECTED

Coastal marine waters serve a wide variety of exceptionally important human uses. Many of these uses produce high local benefits such as the yield of shellfish and recreational activities. Others involve regional benefits or the global unity of the marine system, since local events influence, and are influenced by, water quality at distant points. Many of the human uses of marine waters are directly dependent upon the nature and quality of the biological, chemical, and physical systems present. Efforts to protect and enhance these uses will be limited principally by our ability to understand and protect the environmental conditions which are essential for the biota.

Water quality criteria for marine aquatic life and wildlife define the environmental requirements for specified uses. Five of these are discussed in this Section, namely, maintenance of the ecosystem; fisheries; aquaculture; wildlife protection; and waste disposal. These are not sharply separable, but the water quality requirements for each use are briefly summarized. The effects of transportation, harbor development, dredging and dumping of spoils have also been considered in developing the recommendations.

NATURE OF THE ECOSYSTEM

Many of the principal human uses of marine waters depend upon successful maintenance and enhancement of the existing ecosystems or, in a few circumstances, upon creating and continuing new and artificial ecosystems for specific purposes. The ecosystem includes all of the biological and non-biological (geological, physical, and chemical) components of the environment and their highly complex interactions. Studies of ecosystems must include all that is within the body of water as well as the imports to and exports from it. Research in such situations has shown that the biotic elements include producers of organic material, several levels of consumers, and decomposers. In the least complex situation, these act at rates controlled by the abiotic factors to transfer energy and recycle materials. In those aquatic environments which continuously or intermittently exchange large quantities of energy or materials with other parts of the total global system, understanding and management become more difficult. In the marine environment imports and exports continually occur from coastal runoff, tidal action, oceanic currents, meterological actions, and exchanges with adjacent water bodies or with the benthos and atmosphere. These exchanges are only partially understood, but it is clear that each marine site is connected intimately to the rest of the oceans and to total global mechanisms.

The estuaries are in many ways the most complicated and variable of aquatic ecosystems. Materials carried from the land by rivers vary in quantity and quality, sometimes with strong seasonal patterns of high biological significance. Tidal oscillations cause vigorous reversals of flow. Inherent hydrographic patterns can lead to accumulation of materials and to upstream transport from the point of addition. Dense urban populations on the shores of estuaries produce large amounts of waste, and engineering projects have changed the boundaries and flows of water courses. The biologically rich estuaries are the most variable and the most endangered part of the marine environment.

In each environment the existing characteristics of the system have been produced by dynamic interaction among the components, forces, and processes present. Some of these are small or transitory, but others are massive and enduring. If any one of these forces or processes is changed, a new balance is produced in the system. Relative stability, therefore, results from the balancing of forces, not the absence. The biota are the product of evolution, and each ecosystem contains those species and communities which have adapted to the specific environment over a long period of time and which are successful in that environment. Drastic and rapid modification of the environment, as by pollution, may eliminate some species and encourage others in ways which can reduce the value of the ecosystem for man's use or enjoyment.

Effects of Water Quality Change on Ecosystems

The introduction of a chemical compound or a change in the physical environment may affect a natural marine ecosystem in many ways. In coastal waters undisturbed for long periods of time, the ecosystem has adjusted to the existing conditions. The system is productive, species are diverse, the biomass is high, and the flow of energy is comparatively efficient. The addition of pollutants to such a system might:

- reduce the input of solar energy into the ecosystem;
- increase the input of organic matter and nutrients which might stimulate the growth of undesirable species;
- reduce the availability of nutrients by increased sorption and sedimentation;
- create intolerable physical extremes for some organisms, as by the addition of heat;
- kill or reduce the success of individual organisms, as by lethal toxicity or crippling with oil;
- eliminate species by adding a toxic material or making an essential element unavailable;
- interfere with the flow of energy from species to species, as by a chemical that interferes with feeding behavior;
- reduce species diversity in the system;
- interfere with regenerative cycling by decomposers;
- decrease biomass by reduction of abundant species or disruption of the processes of ecosystems;
- increase biomass by removing important consumers allowing runaway production of other species.

All of these may involve changes in production and lowered human usefulness of the system. These are examples; additional effects can occur. The specific impacts of pollution at a site can be determined only through long-term study of that portion of the ocean and the changes that occur.

It is clear that man, through his numbers and his actions, is having increasingly pronounced effects on organisms, populations, and entire ecosystems. Many people willingly accept the consequences of advanced technology that are markedly deleterious, but most people become alarmed when an entire large ecosystem undergoes transformation. When society recognizes that catastrophe threatens due to its carelessness, it seeks to rearrange its demands on such ecosystems in ways that can be accommodated within the inherent capacities of the system. To provide adequate answers we need understanding of ecosystems, since knowledge at the species and population levels, however defined, will be too limited in scope to answer the questions that arise at the more highly organized level of the ecosystem.

The study of the effects of pollution on ecosystems may be undertaken by considering pollution as an additional stress on the mechanisms that keep ecosystems organized. Unless the living parts of an ecosystem are already under stress, the early effects of the introduction of toxic pollutants may contribute to the extinction of particularly susceptible species leaving the more resistant forms in a less diverse community. In communities already under stress, relatively low levels of pollution may cause the disruption of the communities.

Estuaries and intertidal regions are naturally exposed to stressful conditions. In the estuaries the ebb and flow of the tide and the fluctuating freshwater flow create changes in salinity on various time scales ranging from hourly to seasonally. In the intertidal zone the normal inhabitants are exposed to air during part of each tidal cycle. They are also subjected to vigorous wave actions on exposed beaches and headlands. Unique assemblages of organisms have evolved which manage to survive these rigorous conditions if waters remain unpolluted.

Pollutants are commonly released into such aquatic ecosystems of high natural variation in their nonliving components, and the rate of pollutant discharge usually varies from time to time. The immediate effect of these conditions is that at any fixed point in the habitat the concentration of a pollutant varies markedly with time, but not in such a way that a community can adapt itself to these variations. The result is that short-lived opportunistic species are likely to be favored in areas subject to variable aquatic pollution.

Any single toxicant may be equally virulent towards long-lived or short-lived species in the normal aquatic community. Except at outfalls where toxicants reach lethal concentrations, as in continuous discharges in stable environments, toxicants act discontinuously through time. Where water mass instabilities are such that poisonous concentrations occur on the average of once a week, for instance, it is possible for organisms with much shorter life spans to flourish briefly with large population fluctuations. Where they occur once a month, a community may evolve rapidly through a successional sequence involving a few longer-lived organisms before the next toxic concentration occurs. Where lethal dosages are as infrequent as once a year, the succession may go to the stage of some fish of medium life span, particularly if access to the area is relatively free. Because of the fluctuations with time, the community nearest an outfall is most primitive from a successional viewpoint, and as distance from the outfall increases, there is a successional gradient toward the usual climax community of an unpolluted environment.

Evaluation of the effects of pollution or of other environmental changes on the ecosystem involves studies of biological production, species diversity, energy flow, and cycling of materials. The process may be complicated by massive imports and exports at any one site. Although pathways of energy flow and efficiencies are not yet completely understood, they offer a unifying approach to these problems such as proposed by Odum (1967,[12] 1971[13]).

Species diversity is a useful attribute of biological systems. Diversity is affected by a number of factors as evidenced by the papers presented at a symposium on Diversity and Stability in Ecological Systems (Brookhaven National Laboratory 1969),[10] as well as other symposia (American Society of Civil Engineering and Stanford University 1967,[9] Olson and Burgess 1967,[14] NAS-NRC Committee on Oceanography 1970,[11] Royal Society of London 1971[15]). Some success has been achieved in the use of diversity measurements, however, and their potential for future use is high. (See the discussion of Community Structure in Section III on Freshwater Aquatic Life and Wildlife and in Appendix II-B.)

There are potentials for managing additions to coastal ecosystems in ways that benefit human uses. These are as yet poorly understood, and efforts to utilize waste heat, nutrients and other possible resources are primitive. Such possibilities merit vigorous exploration and, eventually, careful application.

FISHERIES

Major marine and coastal fisheries are based upon the capture of wild crops produced in estuaries, coastal waters, and oceans. The quantity and quality of the available supply of useful species are controlled by the nature and efficiency of the several ecosystems upon which each species depends for its life cycle. Shad, for instance, depend upon freshwater areas at the head of estuaries for spawning and for survival as eggs and larvae, open estuaries for the nutrition of juveniles, and large open coastal regions for growth and maturation. As do many other species, shad migrate over large distances. Serious pollution at any point in the lower river, the estuary, or the inshore ocean might, therefore, break the necessary patterns and reduce the fishery.

Estuaries have exceptional usefulness in support of fisheries. At least three quarters of the species in the commercial and recreational fisheries of the nation are dependent upon the estuarine ecosystem at one or more stages of their life history. Estuaries are used as obligatory spawning grounds, nursery areas, havens from parasites and predators, and as rich sources of food because of high productivity.

American fisheries exploit several levels of the coastal ecosystem. We do not utilize the plants, the producers, directly as food or in commerce except for a comparatively small harvest of kelp and other seaweeds. The primary consumers, however, are extensively utilized. These include oysters, clams, mussels, and vast quantities of filter-feeding fish such as sardines, anchovies, menhaden, and herring. Second and third level consumers, which are less abundant but frequently more desired than plankton feeders, include most of our sports fish and major commercial species such as tuna, striped bass, cod, halibut, and sea trout, as well as squid, sharks, and other species which hold potential for increased future use.

Pollutants can be detrimental to fisheries by reducing desired species through direct mortality from toxicity, smothering, intolerable heat, or other killing changes. Reduction may also occur when a pollutant has a sublethal stressing effect that significantly interferes with feeding, movement, reproduction, or some other essential function. Pollution has an indirect deleterious effect when it increases predators or parasites, reduces food organisms or essential consorts, or damages the efficiency of the ecosystem functions pertinent to the species in question. Consideration of all these occurrences must enter into efforts to protect and enhance fisheries.

Pollutants also damage marine organisms by imparting characteristics that make them unacceptable for commercial or recreational use. Economic loss has resulted from flesh tainting of fish and shellfish by oil, phenolics, and other materials affecting taste, flavor, or appearance. DDT and other persistent organics, applied on land, have accumulated in fish to levels that exceed established standards for acceptable human food. Heavy metals, e.g., mercury, can reach levels in fish several thousand times the concentration in the ambient water, destroying the economic value of the organisms involved.

More than 90 per cent of the American commercial catch and virtually all of the sport fish are taken from the estuaries and continental shelf. The total yield is difficult to estimate, involving as it does migratory species, catches by both foreign and domestic vessels, and recreational fisheries which are only partially measured. Stroud (1971)[26] estimated that the estuarine-dependent fishery of the Atlantic coast yields 535 pounds per acre of estuary for a total annual yield of 6.6×10^9 lbs. He concludes that shrinking of estuaries by filling or other destruction would reduce the yield by a directly proportional quantity. Further, he predicts that reduction of the productivity of estuaries by pollution would also produce a proportional decrease in fish production. The U.S. commercial fisheries of largest volume, in order of decreasing harvest, include menhaden, salmon, shrimp, crabs, herring, and oysters (Riley 1971).[24] The most valuable commercial harvests include shrimp, salmon, lobsters, crabs, menhaden, oysters, clams, flounders, and scallops (Riley 1971).[24]

The estuaries, as recipients of wastes both from rivers entering them and cities and industries along their shores, are obviously more immediately susceptible to pollution damage than any other part of the marine system (Clark 1967,[18] American Society of Civil Engineering and Stanford University 1967,[16] U.S. Dept. of Interior 1969,[27] and U.S. Dept. of Interior, Fish and Wildlife Service 1970[28]). Although the vulnerability of such inshore bodies of water to physical and chemical damage is exceptional, the open waters along the coast are also subject to damage from the use of these waters for waste disposal. Approximately 250 waste disposal sites are in use along the coast of the United States, and 48 million tons of wastes are estimated to have been dumped in 1968 (Council on Environmental Quality 1970).[19] These dumped wastes included dredge spoils, industrial wastes, sewage sludge, construction and demolition debris, solid wastes, and explosives (see pp. 278–283 of this Section for a more extended discussion of dumped wastes). Increased populations and technological concentration along the coasts, with simultaneous resistance to the use of land, rivers, and estuaries for disposal has stimulated proposals to increase the use of oceanic areas as receivers of wastes.

The effects of such coastal disposal on fisheries are not yet clearly established. Bechtel Corp. (1969)[17] has suggested that continued expansion of waste disposal along the Atlantic coast at the present rate of increase may, in about 30 years, significantly reduce the quality of water over the continental shelf by increased suspended solids, phosphate or nitrate enrichment, oxygen demand, heavy metals, or simultaneous effects from all of these. Preliminary studies of the effects of dumping of sewage sludge and dredging spoils from the metropolitan New York area indicate that an area of about 20 square miles has been impoverished by reduction of normal benthic populations; and indirect effects may be far more extensive (Pearce 1970).[23]

More general approaches to disposal of wastes in ocean waters have been presented by Foyn (1965)[20] Olson and Burgess (1967)[22], NAS-NRC Committee on Oceanography (1970)[21] and the Royal Society of London (1971).[25] Some discernible and disturbing changes in coastal waters are documented that prove the urgent need for better understanding of pollution effects at the edge of the oceans. The limitation that must be placed upon any such releases must be learned and put to use quickly, and we should proceed carefully while we are learning.

Fisheries provide useful indications of the biological health and productivity of marine waters. Continuous high yield of a harvestable crop of indigenous fish or shellfish free of toxicants or pathogens is an indication that water quality is satisfactory, that the environmental conditions are favorable for the total biological community, and that no contaminant is present in sufficient quantity to destroy major components of the ecosystem. Fisheries production statistics can thus serve as a sensitive indicator of environmental quanity.

Specific criteria for categories of pollutants will be given in subsequent parts of this Section. The general requirements for water quality in relation to successful fisheries include:

- favorable, not merely tolerable, environmental conditions at every location which is required in the life history of each species: this places special value on water quality of estuaries which are obligate environments for many species during at least some portion of their life cycles;
- freedom from tainting substances or conditions where useful species exist, including elements and compounds which can be accumulated by organisms to unacceptable levels;
- absence of toxic conditions or substances wherever useful species occur at any time in their life history;
- absence of sublethal deleterious conditions which reduce survival and reproductive success;
- water sufficient to maintain the health of the biological systems which support useful species;
- absence of environmental conditions which are exceptionally favorable to parasites, predators, and competitors of useful species.

MARINE AQUACULTURE

Although often considered a new approach to the world food problem, aquaculture is an ancient practice in many parts of the world. In the Orient, aquatic organisms have been successfully cultivated for centuries, usually with rather primitive and empirical techniques, but nevertheless with impressive success.

The annual world production of food through aquaculture has recently been estimated at over four million metric tons, about 6.5 per cent of the total world fish landings. Although this is derived largely from fresh water, and open-ocean maraculture is in its infancy, an unknown but significant fraction of the production is brackish-water organisms taken from estuarine systems. The distinction between freshwater and marine aquaculture is quite artificial. Because the principles, techniques, potentials, and environmental requirements for growing organisms in either fresh or salt water are much the same, the distinction is also unnecessary for the purposes of the present discussion, except as noted below.

It is difficult to assess the potential yield from marine aquaculture, dependent as it is on a primitive art undergoing rapid technological development. The introduction of present methods into new, undeveloped parts of the world could at least double the present harvest within the next decade. Judging from the experience in agriculture and terrestrial animal husbandry, much greater increase in yields should presumably be possible with advances in such fields as genetic selection and control, nutrition, habitat management and elimination or control of disease, predation, and competition. It is not inconceivable that the yield from aquaculture might one day surpass that from the harvest of wild, untended stocks of aquatic organisms. Further, since only the most desirable species are selected for aquaculture, both the economic and nutritional value per pound of cultivated organisms greatly exceeds that of the average fishery product. In the United States, expanded recent interest in coastal aquaculture will hopefully produce new techniques, products, and quantities, although economic feasibility has been difficult to achieve thus far.

Although no firm distinction can be drawn, it is convenient to think of most forms of marine aquaculture in one of two categories that will be referred to here as *extensive* and *intensive* culture. In extensive culture, animals are reared at relatively low densities in large impoundments, embayments, or sections of estuaries, either natural or man-made. The impoundments may be closed off or open to the sea, depending upon the desired degree of control, but even those that are enclosed must be located near a source of seawater so that the water may be exchanged frequently to prevent stagnation and to regulate such factors as temperature and salinity. Such exchanges are accomplished by tidal action or by pumping.

The cultivated animals may be stocked or may consist of natural populations that enter the system as larvae or juveniles. They are usually not fed but subsist on natural foods that grow in the area or are carried in with the outside seawater.

Extensive aquaculture systems are most common in the undeveloped parts of the world (e.g., Southeast Asia) where large areas of coastal mangrove swamps, marshes, and estuaries are available and are not presently in use or demand for other purposes. For example, it has been estimated that there are over six million acres of mangrove swamps in Indonesia alone that would be suitable for some form of fish farming.

In such coastal impoundments, milkfish, mullet, shrimp, and other free-swimming species are grown. In the more open situations such as embayments and arms of estuaries, non-fugitive organisms are cultivated. The oldest and most highly-developed form of marine aquaculture practiced in the United States and Europe, that of oyster culture, falls into this category. Seaweed culture in Japan and China is another interesting example of this general approach to aquaculture.

Yields from extensive aquaculture range from a few hundred pounds to, at best, about one ton per acre per year. Little, in some cases almost no, capital investment is required, and it is not a labor-intensive form of enterprise. One or two unskilled laborers can manage 100 acres or more of shrimp or milkfish ponds in Malayasia or the Philippines except during stocking and harvesting operations. This is normally a highly profitable form of business to the culturist and, despite the modest yields, extensive aquaculture is capable of making a significant contribution to the protein nutrition of many of the undeveloped parts of the world.

Intensive aquaculture makes use of flowing-water systems using flumes or raceways and is best typified by trout and salmon hatcheries that have been operated successfully in the United States and Europe for many years and have now reached a relatively high level of technical sophistication. Although originally designed to produce fish to be stocked in natural waters to enhance commercial or sports fishing, such systems are now being increasingly used for the production of fish to be marketed directly as food. Such systems were originally developed and used exclusively for rearing freshwater species, but they are now also finding application in saltwater areas for the production of marine or anadromous species.

A variation of the raceway system of intensive aquaculture is that of floating cage culture in which the animals are held in nets suspended by a floating wooden framework. These may be moored in estuaries or other protected arms of the sea, where they are exposed to strong tidal currents.

A common feature of the various kinds of intensive aquaculture is that the animals are grown closely packed at extremely high densities and depend upon the flow of large volumes of water over and around them to provide oxygen and carry away wastes. When feasible, the animals are fed artifically on prepared, pelletized food. The entire system must be carefully controlled and monitored.

Intensive aquaculture systems for the commercial production of food are in an early stage of development and have yet to prove themselves as profitable and reliable for marine species. Rapid progress is being made in this area, however, particularly in highly developed parts of the world where technological skill is available, where coastal marine areas are scarce and in high demand, and where the price of luxury seafoods is escalating. Various species of molluscs, crustaceans, and finfish are now being grown in this way, and many more are likely candidates as soom as fundamental aspects of their life history and nutrition are mastered.

The yield from intensive aquaculture per unit of area in which the organisms are grown is ecologically meaningless (as is that from a cattle feed-lot, for example) but amounts to as much as hundreds of tons per acre. More realistically, the yield from such systems may be expressed per cubic foot per minute of water flowing through it, which is usually the limiting factor.

In contrast to extensive aquaculture, intensive systems usually require high capital outlay and have a relatively high labor demand. Profits or losses are determined by small differences in the costs of food, labor, marketing, and the demand for the product.

Both extensive and intensive forms of aquaculture are heavily dependent on high quality water to sustain them. Neither is independent of the adjacent coastal marine environment. Extensive pond culture may be semi-autonomous, but as explained above, the water must be occasionally and sometimes frequently exchanged. Intensive aquaculture systems are vitally dependent on a continuous large supply of new seawater. Because of the large investment and, at best, small margin of profit, and because of the dense populations of animals maintained at any one time, intensive aquaculture represents a far greater risk.

Freshwater aquaculture systems, if strategically located near an adequate source of underground water, may be largely independent of man's activities and relatively free from the threat of pollution. This, unfortunately, is never quite true of marine aquaculture. The contiguous oceans of the world circulate freely, as do the substances man adds to them. While water movements may be predicted on large geographical and time scales, they are quite unpredictable on a local and short-term basis. An embayment or estuary whose shores are uninhabited and which may suffer no ill effects from the surrounding land may suddenly become infused with materials added to the water hundreds of miles distant and carried to the scene by winds, tides, and coastal currents. In this sense, marine aquaculture is not only more vulnerable to change than freshwater culture, but the dangers are also far less predictable.

Application of Water Quality to Aquaculture

The various toxic or otherwise harmful wastes that man adds to the coastal marine environment affect cultivated organisms much the same as they do the natural populations of the same species. These are discussed in detail elsewhere and need not be repeated here. In general the deleterious effects of wastes on organisms that are used as food by man are: (1) to kill, injure, or interfere with the growth or other vital functions of the organisms, or (2) to become concentrated in the organisms to such an extent as to render them unfit for human consumption by exceeding public health standards or by making them unpalatable. In the latter case, this may occur with no apparent accompanying impairment of the organism.

Certain aspects of aquaculture, particularly the intensive forms of culture described above, are particularly sensitive and vulnerable to various kinds of pollution—more so than their freeliving counterparts in nature. These are enumerated and discussed briefly below.

- The carrying capacity of intensive aquaculture systems is based on the flow of water and its supply of oxygen. If the concentration of oxygen in the water suddenly decreases due to an organic overload, a temperature increase, or other external causes, it may be inadequate to support the cultivated animals.
- Captive organisms cannot avoid localized unfavorable conditions (e.g., oxygen, temperature, turbidity) as can free-swimming natural populations.
- Many organisms can tolerate alterations in their environment if they are allowed to adapt and become acclimated to such changes slowly. Cultivated organisms may be, and often are subjected to sudden changes in water quality and cannot endure the initial shock, while the free-swimming natural populations can enter an affected area slowly and cautiously and allow themselves to adapt to the altered conditions.
- Cultivated organisms, particularly in the densely-crowded conditions of intensive aquaculture, may be and perhaps always are under rather severe physiological stress. Artificial diets are often incomplete or otherwise unbalanced. Unnaturally crowded living conditions may cause hormonal or other biochemical imbalance. The animals may already suffer the effects of poor water quality from their own pollutants. They are therefore particularly susceptable and vulnerable to any additional deterioration in water quality that may increase their stress condition.
- Disease is a spectre that perpetually haunts the aquaculturist. Virtually impossible to avoid or eliminate in any open system, it is usually, at best, held in check. Again, the additional stress caused by a deterioration in water quality, while not fatal in itself, may lower the resistance of the cultivated animals to epidemic disease.
- Artificially-fed cultivated organisms may be no less susceptible to accumulation of wastes, although intensively cultivated organisms that are fed entirely on an artificial diet would appear to have one advantage over natural populations of the same animals living in polluted waters. Many toxic substances such as chlorinated hydrocarbons may reach toxic or unacceptable levels in larger organisms because of concentration and amplification at each successive step in the food chain that ultimately supports the animal in question. However, there is increasing evidence that these substances also enter fishes and other organisms directly from solution in the water, across respiratory or digestive membranes. Such direct absorption of toxic material may in some cases exceed the quantities ingested and assimilated with food.

Therefore, the general recommendations for the quality of water for use in culture include: (1) continuously adequate control of those materials and conditions which are required for good health and efficient production of the cultured species; (2) absence of deleterious chemical and physical conditions, even for short or intermittent periods; (3) environmental stability; and (4) prevention of introduction of diseases that attack the organisms under culture. The specific requirements for each culture effort must be with reference to the species involved, the densities desired, and the operational design of the culture system.

MARINE WILDLIFE

Marine wildlife for the purposes of this Section is defined as those species of mammals, birds, and reptiles which inhabit estuaries or coastal and marine waters for at least a portion of their life span. The fish, invertebrates, and plankton that constitute the food webs upon which these species depend are not, therefore, considered to be wildlife in this context. The recommendations for marine wildlife, however, necessarily include all recommendations formulated to protect the fish, invertebrate, and plant communities, because wildlife can be adequately protected only if the diversity and integrity of food webs are maintained. Moreover, the recommendations must protect wildlife from pollutants that are relatively persistent in the environment, transported by wind or water currents, and concentrated or recycled in the food webs. Because of trophic accumulation, birds and mammals that occupy the higher trophic levels in the food web may acquire body burdens of toxicants that are lethal or that have significant sublethal effects on reproductive capacity, even though the concentrations of these substances in the water remain extremely low. Pollutants of concern or of potential concern are the radionuclides, heavy metals, chlorinated hydrocarbons, and other synthetic chemicals that are relatively resistant to biological and chemical degradation.

Recommendations to protect wildlife dependent upon freshwater ecosystems may in general also apply to estuaries. This is particularly true for protection of food and shelter for wildlife, pH, alkalinity, light penetration, settleable substances, and temperature. These are discussed in Section III on Freshwater Aquatic Life and Wildlife.

Marine and coastal waters constitute major sinks for persistent pollutants. Accumulation rates and steady-state levels are complex functions of input, rates of degradation, and rates of deposition in the sediments. As yet no research programs have measured accumulation rates of pollutants in coastal waters or determined whether steady-state concentrations have already been attained.

Current knowledge of the partition coefficients among concentrations in water, in sediments, and in tissues of representative species in food webs is at best fragmentary. It is assumed, however, in the evaluation of water quality that the distribution and concentration of gradients of a pollutant in an aqueous ecosystem satisfy thermodynamic equilibria requirements. The pollutants considered here are not essential to physiological functions, and do not require energy to maintain the concentration gradients. Thus the chlorinated hydrocarbons are concentrated in the lipid pools of organisms from ambient water but will not accumulate indefinitely. Rather, under equilibrium conditions, these pollutants will also be lost to ambient water, particulate matter, and sediments in satisfying thermodynamic requirements. Because the internal environments of birds and mammals are more isolated from the ambient environment than those of invertebrates and most fish, equilibrium concentrations of pollutants, particularly the chlorinated hydrocarbons, may be substantially higher.

Theoretically, therefore, measurements of pollutant concentrations in one component of an ecosystem are sufficient to indicate the level in the system as a whole when the partition coefficients among water, suspended particulate and organic material, sediments, lipid pools, surface films, and the atmosphere are known. The methodologies for measuring pollutant concentrations in sea water are as yet imperfect, and very few good measurements have so far been made. Consequently it is not practical at present to make recommendations for the relatively persistent organic pollutants based upon water concentrations, especially when partition coefficients are not known. Residue concentrations in fish are more easily determined and can more readily be associated with harm to bird and mammal populations that consume them. Recommendations for the toxic organic compounds that are trophically accumulated by marine wildlife are therefore based upon concentrations in fish.

It cannot be assumed that there is a level or concentration in the ecosystem as a whole of pollutants which are mutagens or teratogens that causes no effect on any of the wildlife species. The chlorinated dibenzo-p-dioxins are highly toxic to developing embryos (Verret 1970)[71] and are contaminants in compounds prepared from chlorinated phenols, including the herbicide 2,4, 5-T (Verrett 1970)[71] and the widely used fungicide pentachlorophenol (Jensen and and Renberg 1972).[53] The closely related chlorinated dibenzofurans are contaminants in some PCB preparations (Vos and Koeman 1970,[74] Vos et al., 1970,[75] Vos *in press* 1972).[72] Embryonic mortality in birds is induced by these or other derivatives of PCB (Peakall et al., *in press* 1972,[59] Vos *in press* 1972).[72] For the present time the chlorinated dibenzofurans are included with PCB in the recommendations. When environmental mutagens and teratogens affect only relatively few individuals of a population, it is assumed that these will be eliminated by natural selection without harm to the species as a whole.

For other pollutants which affect specific enzyme systems or other physiological processes but not the genetic material or embryological development, it is assumed that there are levels in the environment of each below which all organisms are able to function without disrupting their life cycles. Manifestations of physiological effects, such as a certain amount of eggshell thinning or higher level of hormone metabolism, might be detectable in the most sensitive species. If environmental levels increase, the reproductive capacity of the most sensitive species would be affected first. The object of the recommendations presented is to maintain the steady-state concentrations of each pollutant below those levels which interfere with the life cycles of the most sensitive wildlife species. Input should not therefore be measured only in terms of concentrations of each pollutant in individual effluents, but in relation to the net contribution to the ecosystem. At the steady-state level, the net contribution would be zero, with the total input equal to the sum of degradation and permanent deposition in the sediments.

Bases For Recommendations

Recommendations based upon pollutant concentrations in fish must take into account the individual variation in residue concentration. The distribution is usually not Gaussian (Holden 1970;[51] Anderson and Fenderson 1970;[30] Risebrough et al. *in press* 1972),[65] with several individual fish in a sample frequently containing much higher residue concentrations than the majority. Fish samples should therefore consist of pooled collections. Samples as large as 100 fish may not be sufficient to determine mean concentrations of a pollutant with a precision of 10 per cent (Risebrough et al. *in press* 1972).[65] Practicality, however, frequently dictates against sample sizes of this magnitude, and samples consisting of 25 or more fish are suggested as a reasonable compromise.

Radionuclides

Recommendation

In the absence of data that would indicate that any of the radionuclides released by human activities are accumulated by wildlife species, it is recommended that the recommendations established for marine fish and invertebrates apply also to wildlife.

Heavy Metals

The results obtained during the baseline study of the International Decade of Ocean Exploration (IDOE) in 1971–72 have failed to indicate any evidence of pollution by heavy metals, including mercury and cadmium, above background levels in marine species (Goldberg 1972).[45] The results, suggested, however, local patterns of coastal contamination. The heavy metal analyses carried out to date of tissues of several species of petrels, strictly pelagic in their distribution (Anderlini et al. 1972);[32] and of coastal fish-eating species such as the Brown Pelican, *Pelecanus occidentalis*, (Connors et al. *in press* 1972a);[40] and the Common Tern, *Sterna hirundo* (Connors et al. *in press* 1972b)[41] have confirmed this conclusion.

Recommendation

In the absence of data indicating that heavy metals are present in marine wildlife in concentrations above natural levels, it is recommended that recommendations formulated to protect other marine organisms also apply to wildlife in order to provide protection in local areas.

Polychlorinated Biphenyls (PCB)

Evidence is accumulating that PCB does not contribute to the shell thinning that has been a major symptom of the reproductive failures and population declines of raptorial and fish-eating birds. Dietary PCB produced no shell thinning of eggs of Mallard Ducks (*Anas platyrhynchos*) (Heath et al. *in press* 1972),[49] nor did PCB have any effects on eggs of Ring Doves (*Streptopelia risoria*) (Peakall 1971).[58] A PCB effect could not be associated with the thinning of Brown Pelican (*Pelecanus occidentalis*) eggshells (Risebrough *in press* 1972).[62] PCB may increase susceptibility to infectious agents such as virus diseases (Friend and Trainer 1970).[44] Like other chlorinated hydrocarbons, PCB increases the activity of liver enzymes that degrade steroids, including the sex hormones (Risebrough et al. 1968;[64] Street et al. 1968).[67] The ecological significance of this phenomenon is not clear.

Because laboratory studies have indicated that PCB, with its derivatives or metabolites, causes embryonic death of birds (Vos et al. 1970;[75] Vos and Koeman 1970;[74] Vos *in press* 1972;[72] Peakall et al. *in press* 1972[59]) and because exceptionally high concentrations are occasionally found in fish-eating and raptorial species (Risebrough et al. 1968;[64] Jensen et al. 1969[52]), it is highly probable that PCB has had an adverse effect on the reproductive capacity of some species of birds that have shown population declines.

Median PCB concentrations in whole fish of eight species from Long Island Sound, obtained in 1970, were in the order of one milligram per kilogram (mg/kg) (Hays and Risebrough 1972),[47] and comparable concentrations have been reported from southern California (Risebrough 1969).[61] On the basis of the high probability that PCB in the environment has contributed to the reproductive failures of fish-eating birds, it is desirable to decrease these levels by at least a factor of two (see Section III on Freshwater Aquatic Life and Wildlife pp. 175–177).

Recommendation

It is recommended that PCB concentrations in any sample consisting of a homogenate of 25 or more whole fish of any species that is consumed by fish-eating birds and mammals, within the same size range as the fish consumed by any bird or mammal, be no greater than 0.5 mg/kg of the wet weight.

In the absence of a standardized methodology for the determination of PCB in environmental samples, it is recommended that estimates of PCB concentrations be based on the commercial Aroclor® preparation which it most closely resembles in chlorine composition. If the PCB composition should resemble a mixture of more than one Aroclor®, it should be considered a mixture for the basis of quantitation, and the PCB concentration reported should be the sum of the component Aroclor® equivalents.

DDT Compounds

DDT compounds have become widespread and locally abundant pollutants in coastal and marine environments of North America. The most abundant of these is DDE [2,2-bis(p-chlorophenyl) dicholoroethylene], a derivative of the insecticidal DDT compound, p,p′-DDT. DDE is more stable than other DDT derivatives, and very little information exists on its degradation in ecosystems. All available data suggest that it is degraded slowly. No degradation pathway has so far been shown to exist in the sea, except deposition in sediments.

Experimental studies have shown that DDE induces shell thinning of eggs of birds of several families, including Mallard Ducks (*Anas platyrhynchos*) (Heath et al. 1969),[48] American Kestrels (*Falco sparverius*) (Wiemeyer and Porter 1970),[77] Japanese Quail (*Coturnix*) (Stickel and Rhodes 1970)[66] and Ring Doves (*Streptopelia risorial*) (Peakall 1970).[57]

Studies of eggshell thinning in wild populations have reported an inverse relationship between shell thickness and concentrations of DDE in the eggs of Herring Gulls (*Larus argentatus*) (Hickey and Anderson 1968).[50] Double-crested Cormorants (*Phalacrocorax auritus*) (Anderson et al. 1969),[31] Great Blue Herons (*Ardea herodias*) (Vermeer and Reynolds 1970),[70] White Pelicans (*Pelecanus erythrorhynchos*) (Anderson et al. 1969),[31] Brown Pelicans (*Pelecanus occidentalis*) (Blus et al. 1972;[36] Risebrough *in press* 1972),[62] and Peregrines (*Falco peregrinus*) (Cade et al. 1970).[37]

Because of its position in the food webs, the Peregrine accumulates higher residues than fish-eating birds in the same ecosystem (Risebrough et al. 1968).[64] It was the first North American species to show shell thinning (Hickey and Anderson 1968).[50] It is therefore considered to be the species most sensitive to environmental residues of DDE.

The most severe cases of shell thinning documented to date have occurred in the marine ecosystem of southern California (Risebrough et al. 1970)[63] where DDT residues in fish have been in the order of 1–10 mg/kg of the whole fish (Risebrough *in press* 1972).[62] In Connecticut and Long Island, shell thinning of eggs of the Osprey (*Pandion haliaetus*) is sufficiently severe to adversely affect reproductive success; over North America, shell thinning of Osprey eggs also shows a significant negative relationship with DDE (Spitzer and Risebrough, *unpublished results*).[78] DDT residues in collections of eight species of fish from this area in 1970 ranged from 0.1 to 0.5 mg/kg of the wet weight (Hays and Risebrough 1972).[47] Evidently this level of contamination is higher than one which would permit the successful reproduction of several of the fish-eating and raptorial birds.

Recommendation

It is recommended that DDT concentrations in any sample consisting of a homogenate of 25 or more fish of any species that is consumed by fish-eating birds and mammals, within the same size range as the fish consumed by any bird or mammal, be no greater than 50 μg/kg of the wet weight. DDT residues are defined as the sum of the concentrations of p,p′-DDT, p,p′-DDD, p,p′-DDE and their ortho-para isomers.

Aldrin, Dieldrin, Endrin, and Heptachlor

Aldrin, dieldrin, endrin, and heptachlor constitute a class of closely related, highly toxic, organochlorine insecticides. Aldrin is readily converted to dieldrin in the environment, and heptachlor to a highly toxic derivative, heptachlor epoxide. Like the DDT compounds, dieldrin may be dispersed through the atmosphere (Tarrant and Tatton 1968,[68] Risebrough et al. 1968).[64] The greatest hazard of dieldrin is to fish-eating birds such as the Bald Eagle (*Haliaeetus leucocephalus*) (Mulhern et al. 1970);[56] the Common Egret (*Casmerodius albus*) (Faber et al. 1972)[43] and the Peregrine (*Falco peregrinus*) (Ratcliffe 1970),[60] which may accumulate lethal amounts from fish or birds that have not themselves been harmed.

These compounds are somewhat more soluble in water than are other chlorinated hydrocarbons such as the DDT group (Gunther et al. 1968);[46] partition coefficients between water and fish tissues can be assumed to be lower than those of the DDT compounds. Equivalent concentrations in fish would therefore indicate higher environmental levels of dieldrin, endrin, or heptachlor epoxide than of DDE or any of the other DDT compounds. Moreover, these compounds are substantially more toxic to wildlife than are other chlorinated hydrocarbon pesticides (Tucker and Crabtree 1970).[69] More conservative recommendations are therefore necessary.

Recommendation

It is recommended that the sum of the concentrations of aldrin, dieldrin, endrin, and heptachlor epoxide in any sample consisting of a homogenate of 25 or more whole fish of any species that is consumed by fish-eating birds and mammals, within the size range consumed by any bird or mammal, be no greater than 5 μg/kg of the wet weight.

Other Chlorinated Hydrocarbon Pesticides

Other chlorinated hydrocarbon insecticides include lindane, chlordane, endosulfan, methoxychlor, mirex, and toxaphene. Hexachlorobenzene is likely to have increased use as a fungicide as mercury compounds are phased out. This compound is toxic to birds and is persistent (Vos et al. 1968).[73] With the possible exception of hexachlorobenzene, recommendations that protect the invertebrate and fish life of estuaries from injudicious use of these pesticides will also protect the wildlife species. In light of the experience with DDT and dieldrin, the large scale use of a compound such as mirex can be expected to have adverse effects on wildlife populations.

Recommendation

It is recommended that the concentration of any of these chlorinated hydrocarbon insecticides, including lindane, chlordane, endosulfan, methoxychlor, mirex, and toxaphene, and of hexachlorobenzene, in any sample consisting of a homogenate of 25 or more whole fish of any species that is consumed by fish-eating birds and mammals, with the size range that is consumed by any bird or mammal, be no greater than 50 μg/kg of the wet weight.

Lead

No data was found to indicate that lead released into the atmosphere through the combustion of leaded gasolines has posed a hazard to wildlife populations or has resulted in an increase in body burdens of lead over background levels. Critical studies, however, have not yet been carried out. Ingestion of lead shot by waterfowl, which often mistake spent lead shot for seed or grit, kills many birds, and the pollution of marshes by lead shot is a serious problem.

Jordan (1952)[54] found that female waterfowl are about twice as sensitive to poisoning as males, and that toxicity varied greatly, depending on species, sex, and quantity and quality of food intake. A corn diet greatly increased the toxicity of lead. A study carried out by Bellrose (1951)[34] indicated that the incidence of lead shot in gizzards of waterfowl averaged 6.6 per cent in 18,454 ducks. Among infected ducks, 68 per cent contained only one shot in their gizzards, and only 17.7 per cent contained more than two (Jordan and Bellrose 1951).[55] The incidence of ingested shot appears to increase throughout the hunting season with a subsequent decline afterwards. Most losses of waterfowl due to ingested lead shot are in fall, winter, and early spring (Jordan 1952).[54] Different species show different propensities to ingest shot. Redhead (*Aythya americana*), Canvasback (*Aythya valisneria*) and Ringnecked Ducks (*Aythya collaris*) are prone to ingest shot, while Gadwall (*Anas strepera*), Teal (*Anus* sp.) and Shoveler (*Spatula clypeata*) show a low incidence. Ingestion of one shot does not appear to produce measurable changes in longevity, but six No. 6 shot are a lethal dose to Mallards, Pintail (*Anus acuta*) and Redheads (Wetmore 1919).[76] Cook and Trainer (1966)[42] found that four to five pellets of No. 4 lead shot were a lethal dose for Canada Geese (*Branta canadensis*). On a body weight basis, 6 to 8 mg/kg/day is detrimental to Mallards (Coburn et al. 1951).[39]

Lead concentrations in livers of poisoned birds are of a comparable order of magnitude, ranging from 9 to 27 mg/kg in Canada Geese (Adler 1944),[29] 18 to 37 mg/kg in Whistling Swans (*Olor columbianus*) (Chupp and Dalke 1964)[38] and an average of 43 mg/kg in Mallards (*Anas platyrhynchos*) (Coburn et al. 1951).[39] These levels are 10 to 40 times higher than background, which is in the order of one mg/kg of the wet weight liver (Bagley and Locke 1967).[33]

Lead poisoning in waterfowl tends to occur especially in areas where a few inches of soft mud overlay a hard substrate. In marshes where waterfowl are hunted, the number of lead pellets per acre of marsh bottom is on the order of 25,000 to 30,000 per acre and is frequently higher (Bellrose 1959).[35] 30,000 pellets per acre are equivalent to 0.7 pellets per square foot.

The data examined indicate that the annual loss is between 0.7 per cent and 8.1 per cent of a population estimated to be 100 million birds. Although there is apparently no evidence that a loss of this magnitude has long-term detrimental effects on any species, it is considered unacceptable. Levels of lead shot in the more polluted marshes should therefore be reduced. The ultimate solution to this problem is the production of non-toxic shot.

Recommendation

In order to reduce the incidence of lead poisoning in freshwater and marine waterfowl, it is recommended that: non-toxic shot be used, or that no further lead shot be introduced into zones of shot deposition if lead shot concentrations exceed 1.0 shot per 4 square feet in the top two inches of sediment.

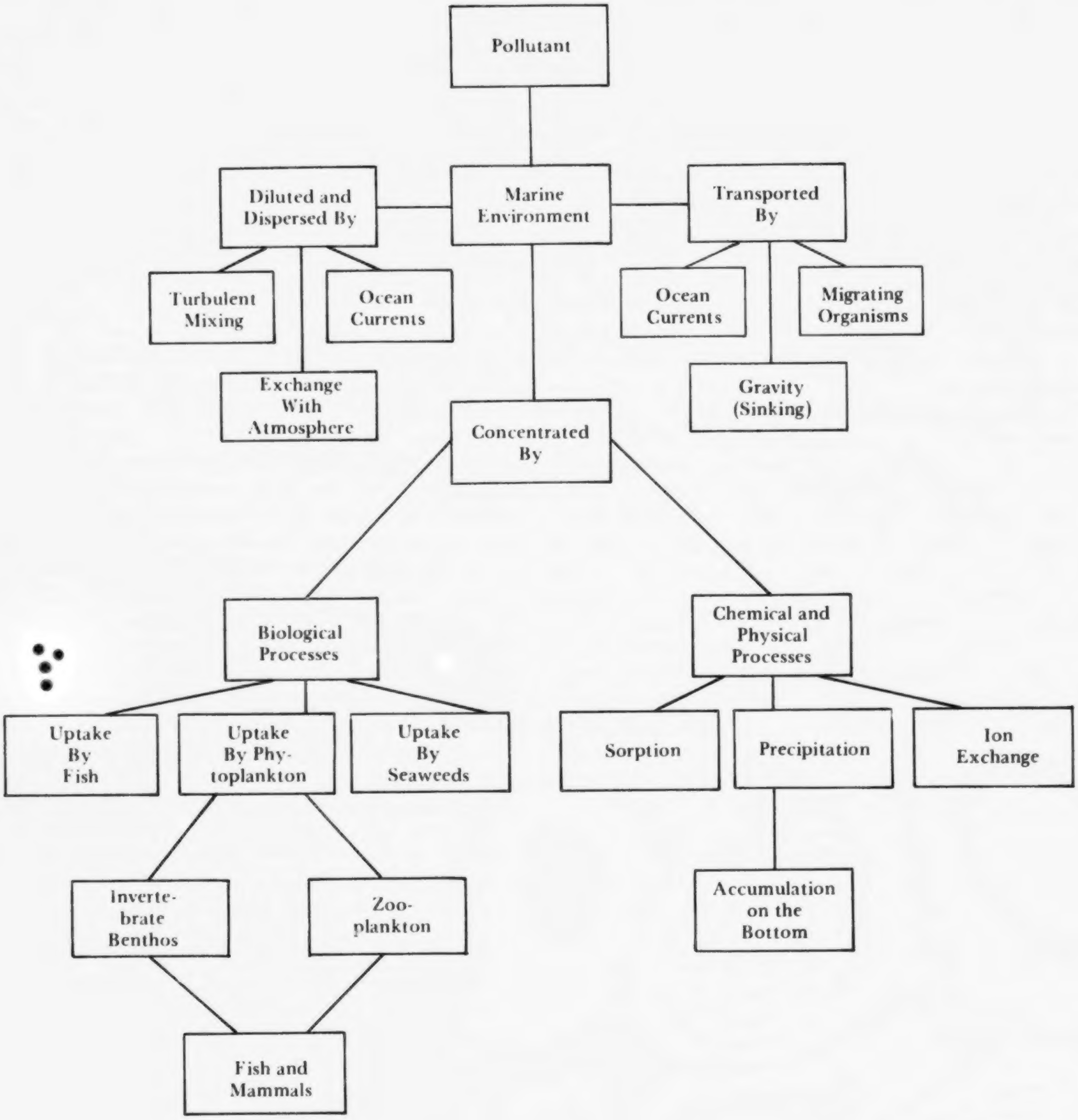
WASTE CAPACITY OF RECEIVING WATERS

When waste disposal to any natural body of water is considered, the receiving capacity of the environment must be taken into account. Waste disposal has been one of the many uses man has required of estuaries and coastal waters. These waters are capable of assimilation of definable quantities and kinds of wastes that are not toxic and that do not accumulate to unacceptable levels. In many locations wastes are being added to these waters at rates that exceed their capacity to recover; and when the rate of addition exceeds the recovery capacity, the water quality deteriorates rapidly. It is essential to understand the local conditions and the processes that determine the fate, concentration, and distribution of the pollutant in order to determine the amount of the pollutant and the rate of disposal that will not exceed the recommended levels.

A simplified diagram of the various processes that may determine the fate and distribution of a pollutant added to the marine environment is presented in Figure IV-1 (Ketchum 1967).[82] The waste material may be diluted, dispersed, and transported by various physical processes, such as turbulent mixing, ocean currents, or exchanges with the atmosphere. It may be concentrated by various biological processes, such as the direct uptake by organisms of a dissolved material in the water, and it may be transferred from organism to organism in various trophic levels of the food web. Additional concentration of the material may occur at the higher trophic levels, particularly if some organ or tissue of the body accumulates the substance, such as DDT or petroleum products that accumulate in the fatty tissues, various metals that may accumulate in the bone or liver, and iodine which accumulated in the thyroid.

Substances can also be concentrated from the environment by chemical, physical, and geological processes such as sorption. Natural waters contain a certain amount of suspended material, and some material added to the water may be sorbed on these particles. In sea water, which already contains in solution most of the known elements, added materials may be precipitated from the water by various chemical reactions. As fresh waters carry pollutants to the sea, the change in salinity causes flocculation of some of the materials suspended in the fresh water and results in their precipitation from the medium. Ion exchange reactions with the various organic compounds dissolved in sea water can also occur.

The average concentration of a given pollutant continuously added to a body of water, will tend to approach a steady state in the system. This concentration is determined by the rate of addition of the pollutant, the rate of its removal or dilution by the circulation, and the rate of its decomposition or removal by biological, chemical, or geo-


Pollutant
Diluted and Dispersed By
Marine Environment
Transported By
Turbulent Mixing
Ocean Currents
Exchange With Atmosphere
Ocean Currents
Migrating Organisms
Gravity (Sinking)
Concentrated By
Biological Processes
Chemical and Physical Processes
Uptake By Fish
Uptake By Phytoplankton
Uptake By Seaweeds
Sorption
Precipitation
Ion Exchange
Invertebrate Benthos
Zooplankton
Accumulation on the Bottom
Fish and Mammals

Ketchum 1967[82]

FIGURE IV-1—Processes That Determine the Fate and Distribution of a Pollutant Added to the Marine Environment.

logical processes. The average concentration is not always the critical concentration to be evaluated. For example, if bioaccumulation occurs, the amount accumulated in the critical organism should be evaluated, rather than the average concentration in the system as a whole. The processes of circulation and mixing may leave relatively high concentrations in one part of the system and low concentrations in another. The average conditions thus set an upper limit on what can be added to the system but do not determine the safe limit. It is clear, however, that a pollutant might be added to a body of water with vigorous circulation at a rate that could result in acceptable water quality conditions, while the same rate of addition of the pollutant to a sluggish stream might produce unacceptable levels of contamination. Thus, the characteristics of the receiving body of water must be taken into account when evaluating the effects of the pollutant upon the environment.

In a stream, the diluting capacity of a system is relatively easy to determine from the rate of addition of the pollutant and the rate of stream flow. The pollutant is carried downstream by the river flow, and "new" water is always available for the dilution of the pollutant. This is not necessarily true of lakes where the pollution added over a long period of time may accumulate, because only a small fraction of the added pollutant may be removed as a result of flushing by the outflow. In estuaries, the situation is further complicated by the mixture of salt and fresh water, because a pollutant added at a mid-point in the estuary can be carried upstream by tidal mixing just as the salt is carried upstream. The upstream distribution of a conservative pollutant is porportional to the upstream distribution of salt, whereas the downstream distribution of the pollutant is proportional to the downstream distribution of fresh water.

In either lakes or estuaries, the average retention time or the half-life of the material in the system can be used to estimate the average concentration that the pollutant will achieve in the system. In lakes, an estimate of the average retention time can be derived from the ratio of the volume of the lake divided by the rate of inflow (or outflow). When the lake is stratified, only part of the volume of the lake enters into the active circulation, and an appropriate correction must be made. In estuaries and coastal waters, a similar calculation can be made by comparing the volume of fresh water in the estuary with the rate of river inflow. The amount of fresh water in any given sample can be computed from the determination of salinity. In stratified estuaries such as a fjord, only the part of the system that is actively circulated should be taken into account. This may be adequately done by the choice of the appropriate base salinity in computing the fresh water content. Examples of the mean retention time of a few bodies of water calculated as described above are presented in Table IV-1.

Lakes with large volumes superficially appear to have a great capacity to accept waste materials. If the retention time is long, however, this merely means that it takes a long time to build up to steady-state concentration, and it will take a comparably long time to recover from a steady-state concentration once it is achieved. For Lake Superior, for example, it would take 128 years to remove half of the steady-state concentration of a pollutant that had been achieved over 185 years at the average rate of input. Aquatic environments in which the circulation is more rapid will achieve a steady-state concentration of a pollutant more quickly and will also recover more quickly.

TABLE IV-1—Average Retention Times and Half Lives for River Water in the Great Lakes and in Various Estuaries and Coastal Regions

	Surface area mi²	Mean retention time	Half life	Reference
Lake Superior	31,820	185 yrs.	128 yrs.	Beeton (1969)[79]
Lake Michigan	22,420	100 yrs.	69 yrs.	Beeton (1969)[79]
Lake Huron	23,010	30 yrs.	21 yrs.	Beeton (1969)[79]
Lake Erie	9,930	2.8 yrs.	1.9 yrs.	Beeton (1969)[79]
Lake Ontario	7,520	8 yrs.	5.6 yrs.	Beeton (1969)[79]
Continental Shelf				
Capes Cod to Hatteras to 1,000 ft. contour	29,000	1.6–2.0 yrs.	1.1–1.4 yrs.	Ketchum and Keen (1955)[84]
New York Bight	483 to 662	6.0–7.4 days	4.1–5.05 days	Ketchum et al. (1951)[85]
Bay of Fundy	3,300	76 days	52 days	Ketchum and Keen (1953)[83]
Delaware Bay				
high river flow	....	48–126 days	33–87 days	
low river flow				Ketchum (unpublished) (1952)[87]
Raritan Bay				
high river flow	45	15–30 days	10–21 days	Ketchum (1951a,[80] b[81])
low river flow				
Long Island Sound	930	36 days	25 days	Riley (1952)[86]

Nonconservative pollutants are those that change with time by processes which are additional to circulation and dilution. The half-life of these substances in the environment is the product of these processes and the processes of circulation and dilution. For radioactivity, for example, the half-life is the time needed for the normal radioactive decay to dissipate half of the radiation of the material. This is different for each radioisotope and may vary from fractions of a second to centuries. The half-life for the decomposition of the organic matter in sewage in marine systems is probably on the order of days and will be dependent on temperature. The decomposition of sewage, however, releases the fertilizing elements in the organic molecule, and these will persist in the environment. In contrast to these rapid changes, the half-life of the chlorinated hydrocarbon pesticides is probably of the order of 10 years in the marine environment, though this is an estimate and not a direct determination. Heavy toxic metals, which may also pollute the environment, do not decay but persist indefinitely, though their location and forms in the system may change with time.

The greatest pollution danger arises from the addition of persistent materials to those ecosystems with slow circulations. Under these conditions, the waste concentration will increase slowly until a steady-state level is reached. If circulation is more rapid, the system will reach steady-state more quickly, but the concentration for a given rate of addition will be less. If the material is not persistent, the rate of decomposition may be more important than circulation in determining the steady-state concentration. If the products of decomposition are persistent, however, these will accumulate to levels greater than those in the original discharge. Local concentrations, such as can be found in the deeper waters of stratified systems or in trapping embayments, may be more significant than the average concentration for the whole system. In short, the recommendations cannot be used to determine the permissible amount of a pollutant to be added or a rate of addition without detailed knowledge of the specific system which is to receive the waste.

Mixing Zones

When a liquid discharge is made to a receiving system, a zone of mixing is created. In the past, these zones have frequently been approved as sites of accepted loss, exempted from the water quality standard for the receiving water. Physical description, biological assessment, and management of such zones have posed many difficult problems. The following discussion deals with criteria for assuring that no significant damage to marine aquatic life occurs in such mixing zones. Although recent public, administrative, and scientific emphasis has focused on mixing zones for the dispersion of waste heat, other uses of the mixing zone concept are also included in these considerations.

Definition of a Mixing Zone A mixing zone is a region in which an effluent is in transit from the outfall source of the receiving waters. The effluent is progressively diluted, but its concentration is higher than in the receiving waters.

Approach to the Recommendation Mixing zones must be considered on a case-by-case basis because each proposed site involves a unique set of pertinent considerations. These include the nature, quantity, and concentration of the effluent material; the physical, chemical and biological characteristics of the mixing area and receiving waters; and the desired uses of the waters. However, the following general recommendation can be established for the purpose of protecting aquatic life in areas where effluents are mixing with receiving waters:

> *The total time-toxicity exposure history must not cause deleterious effects in affected populations of important species, including the post-exposure effects.*

Meeting the Recommendation Special circumstances distinguish the mixing zone from the receiving waters. In the zone, the duration of exposure to an effluent may be quite brief, and it is usually substantially shorter than in the receiving waters, so that assays involving long periods of exposure are not as helpful in predicting damage. In addition, the concentration of effluent is higher than in receiving waters. Therefore, the development of specific requirements for a specific mixing zone must be based upon the probable duration of the exposure of organisms to the effluent as well as on the toxicity of the pollutant.

The recommendation can be met in two ways: use of a probably-safe concentration requirement for all parts of the mixing zone; or accurate determination of the real concentrations and duration of exposures for important species and good evidence that this time-toxicity exposure is not deleterious. The latter, more precise approach to meeting the recommentation will require:

- determination of the pattern of exposure of important species to the effluent in terms of time and concentration in the mixing zone;
- establishment of the summed effects on important species;
- determination that deleterious effects do not occur.

Complexities in the Marine Environment Some of the problems involved in protecting marine aquatic life are similar to those in lacustrine and fluvial fresh waters and, in general, the recommendations in Section III, pp. 112–115 are applicable to marine situations. There are, however, special complexities in evaluating mixing zones in coastal and oceanic waters. These include:

- the exceptional importance of sessile species, especially in estuaries and near shore, where effluents originate;
- the presence of almost all species in the plankton at some stage in the life history of each, so that they may be entrained in the diluting waters;
- obligate seasonal migrations by many fish and some invertebrates;
- oscillation in tidal currents, mixing mechanisms and in resulting concentrations, dilution rates, and dispersion patterns.

None of these affect the general recommendation, but they do contribute to the difficulty of applying it.

Theoretical Approach to Meeting the Recommendation Any measure of detrimental effects of a given concentration of a waste component on aquatic or marine organisms is dependent upon the time of exposure to that waste concentration, at least over some restricted but definable period of time. For a given species and substance, under a given set of environmental conditions, there will be some critical concentration below which a particular measure of detrimental effects will not be observed, regardless of the duration of exposure. Above the critical concentration, the detrimental effects will be observed if the exposure time is sufficiently long. The greater the concentration of the substance, the shorter the time of exposure to cause a specified degree of damage. The water quality characteristics for mixing zones are defined so that the organisms to be protected will be carried or move through the zone without being subjected to a time-exposure history that would produce unacceptable effects on the population of these species in the water body.

In order to quantify this statement, the following quantities are defined:

T50,C,E = time of exposure of a critical aquatic or marine species to a concentration, C, of a given pollutant, under a constant set of environmental conditions, E, which produces 50 per cent mortality of the critical species.

T0,C′,E = time of exposure of a critical aquatic or marine species to a concentration, C′, of a given pollutant, under a constant set of environmental conditions, E, which produces *no unacceptable* effects on the critical species.

For some pollutants, C and C′ for a given time of exposure may be related by:

$$C' = C - \Delta C_0$$

where ΔC_0 is the amount by which the concentration which produces a 50 per cent mortality must be decreased in order that *no unacceptable* effects of the pollutant on a given critical species will occur. For example, in the case of temperature, it has been shown that at temperatures 2 C below those which produce a 50 per cent mortality, no observable detrimental effects occur. For temperature, then, 2 C is a conservative value of ΔC_0.

For other pollutants, notably chemical toxicants, C′ is related to C by the relationship:

$$C' = k \cdot C$$

where k is the ratio of the concentration at which *no unacceptable* effects occur to the concentration which produces a 50 per cent mortality with both concentrations determined over the same exposure time.

It is difficult to establish with statistical confidence a relationship between T0, C′, E and C′, for a large number of species, by direct laboratory experiments. However, laboratory experiments can be used to determine, for the critical species of the receiving waterbody, the relationship between pollutant concentration and the time period of exposure necessary to produce a 50 per cent mortality. Thus, it is necessary to obtain, by experiment, the form and constants of a function of the pollutant concentration, $f_1(C)$, such that

$$T50, C, E = f_1(C).$$

Conservative estimates of ΔC_0 or of k can be obtained dependent upon decisions as to acceptable effects from additional laboratory studies. Once ΔC_0 or k have been established, the relationship $C' = C - \Delta C_0$, or the relationships $C' = k \cdot C$, depending on the properties of the particular waste materials, can be combined with the above equation relating T50, C, E and C, to produce an equation relating T0, C′, E and C′. That is:

$$T0, C', E = f_2(C').$$

This equation gives the maximum time that a particular species could be exposed to a concentration C′ without resulting in unacceptable effects on the population of this species. The water quality recommendations for the mixing zone are satisfied if, for any organisms carried through the mixing zone with the flow or purposefully moving through the zone, the time of exposure satisfies the relationship

$$1 \geqslant \frac{\text{time of exposure}}{f_2(C')}$$

where C′ is the concentration of a specified pollutant in the mixing zone.

Because, in fact, the concentration in the mixing zone decreases with distance from the point of discharge, and hence organisms carried through the plume will be subjected to concentrations which are continually decreasing with time, a more suitable quantitative statement of water quality characteristics necessary for the mixing zone is:

$$1 \geqslant \frac{\Delta T1}{f_2(C'_1)} + \frac{\Delta T2}{f_2(C'_2)} + \frac{\Delta T3}{f_2(C'_3)} \cdots \frac{\Delta Tn}{f_2(C'_n)}$$

where the time of exposure of an organism passing through the mixing zone has been broken into n increments, ΔT1, ΔT2, ΔT3, etc. long. The organism is considered to be exposed to concentration C'_1 during the time interval ΔT1, to concentration C'_2 during the time interval ΔT2, etc. The sum of the individual ratios must then not exceed unity.

The above theory is applied in the recommendations and examples in Section III on Freshwater Aquatic Life and Wildlife, pp. 112–115, and in the Freshwater Appendix II-A, pp. 403–407.

METHODS OF ASSESSMENT

It is the purpose of this discussion to explain the approaches considered in deriving the recommendations given in this Section. Because the biological effects of a pollutant are manifest in a variety of ways, the specific technique to be used in estimating biological impact must be tailored to each specific problem. For example, acute or lethal toxicity of a given pollutant to a marine species can be evaluated by short-term bioassay in the laboratory designed to determine the concentration of the material which is lethal to half of the selected population in a fixed period of time, commonly four days (LC50-96 hours). The "safe" limit will be much lower than the concentration derived in such a bioassay, and appropriate safety factors must be applied. The safe limit should permit reproduction, growth, and all normal life processes in the natural habitat.

When a pollutant is discharged to the environment at a safe concentration determined in this way, the living organisms are exposed to a chronic, sublethal concentration. Some stages of the life cycle of the species to be protected, such as the eggs or larvae, may be more sensitive than the adult stages. It is sometimes possible to identify the critical life stage which can then be used in a bioassay. Long-term bioassays covering a substantial part of the life cycle of the organism can be conducted in the laboratory to determine chronic sublethal effects of pollutants. Various processes of the organism, such as respiration, photosynthesis, or activity may be used to evaluate sublethal effects. Some longterm chronic effects may be more subtle and more difficult to evaluate under laboratory conditions. Examples of this type include changes in breeding or migratory behavior or the development of a general debility making the organisms more suceptible to disease, predation, or to environmental stresses.

A pollutant in the marine environment may also have an effect on the ecosystem not directly associated with its effect on an individual species. Ecosystem interactions are difficult to assess in the laboratory, and techniques for evaluating them in the field are not completely satisfactory. Such interactions must be considered, however, in applying recommendations to any specific situation.

ACUTE TOXICITIES—BIOASSAYS

Detailed methods for laboratory bioassays are described in Section III, Freshwater Aquatic Life and Wildlife, and can serve as guidlines for application to the marine system. The ability to extrapolate from results of bioassay tests is limited, and the need for safety factors in their application to the environment must be emphasized. The methodologies discussed are illustrative and should be considered as guidelines for meaningful bioassays.

The most important uses of bioassays for evaluating water quality are:

- analysis of the concentration of a specific material in natural waters by means of a biological response;
- detection of toxic substances in organisms used as food for man;
- analysis of the suitability of natural waters for the support of a given species or ecosystem;
- determination of critical toxic levels of substances to selected species;
- evaluation of bio-stimulation effects by materials such as nutrients.

These purposes fall into two general categories: bioanalysis and bioresponse.

BIOANALYSIS

Bioanalysis has been used for many years to measure effects of substances on organisms. These assays may give quantitative measurements, such as weight per volume, or be expressed in arbitrary units defined by the degree of response. They are most valuable when the organism responds to a lower concentration than can be detected by available chemical or physical techniques. Such bioassays require carefully controlled procedures, and organisms and experimental conditions must be standardized. Responses are used that have been shown to have a correlation with the amount of test substance present. Preparation of test materials is rigidly controlled to avoid problems arising from synergists or antagonists administered with the test material. This is difficult and often impossible in the bioassay of materials obtained from the environment.

Bioanalysis has potential in measuring pollutants in materials to be discharged to the environment. For toxic materials, the amount of material relative to the biomass of the test organism must usually be controlled, because most toxicants exhibit a threshold effect. It is usual to determine the concentration of material at which some fraction of the maximal effect (commonly 50 per cent) occurs in a population of known and constant biomass. The fact that far lower concentrations present for a longer time might ultimately produce the same effect does not invalidate this type of assay, because quantitation is obtained by comparison with standard curves. It should, however, be realized that in the presence of detoxification mechanisms, the assay should be conducted for a period of time at which the desired effect (such as 50 per cent inhibition) occurs at the lowest possible concentration.

In assays of materials for which an organism has a natural or induced requirement, it must first be established that of all substances which could be present in the sample, only one can produce the response measured. Second, no substances present should reduce the availability of the material. If the first of these conditions is satisfied, the second can often be approached using a "system of adds" in which a graded series of concentrations of standard material are added to the unknown amount of material in the sample. The intercept of the response curve with the concentration axis is a measure of the amount present in the sample.

If zero response is at a finite concentration, a biologically effective threshold concentration (zero) must be used which has been derived from a separate experimental series in the same medium devoid of unknown amounts of test material.

BIORESPONSE

Bioassays which measure the biological *effect* of a substance or mixture on a single organism or artificial ecosystem can be used to establish water quality criteria, to monitor compliance with standards stated in terms of biological effect, or to measure the relative effects of various materials. Natural processes of equilibration, chemical degradation, and physical adsorption are specifically desired, because it is the biological *effect* rather than the *amount* of test material that is of concern. The observed effect will be determined by the availability of the material, the rate of formation or degradation, and the effect of chemical by-products; and by alterations of the environment caused by addition of the material. Whether conducted in the laboratory or in the field, this type of bioassay is performed on time scales varying from determinations of acute toxicity (commonly 96 hours or less) through determinations of incipient LC50 levels (Sprague 1969,[94] 1971[96]), and on time scales which include multiple generation chronic exposures. Each of these has its own utility and limitations.

Short-term determinations of TLm or TL50 values are primarily of value in comparing toxicities of a number of formulations which have similar modes of action. They are also useful in determining the dilution to be employed in long-term, flow-through exposure and in comparing sensitivities of various life stages of the same organism. In practical terms, each life stage must be considered a physiologically distinct organism with its own particular environmental requirements: immature stages commonly have quite a different habitat and may have different sensitivity.

It has been common practice to use information from acute toxicity studies to establish concentrations tolerable for natural waters. This is done by multiplying the level found in the bioassay by some more or less arbitrary "application factor" (Henderson 1957,[91] Tarzwell 1962[97]). Recently, there have been attempts to establish the application factor experimentally (Mount 1968,[92] Brungs 1969[90]). Application factors are discussed in Section III, Freshwater Aquatic Life and Wildlife, and that discussion is applicable to the marine system. If, in the process of conducting these assays, organisms are periodically removed to an uncontaminated medium, the time of exposure which the organism can withstand and still survive, should it escape the pollutant or should the pollutant degrade rapidly after a single addition, can also be estimated.

Determination of incipient LC50 is a valid measurement of acute toxicity, because the assay is continued until maximum effect is observed at any given concentration (Sprague 1969,[94] 1970,[95] 1971[96]). These bioassays must be conducted under conditions of continuous flow, because the degree of response cannot be limited by the absolute amount of toxicant available in the system or by the relationship between biomass and absolute amount. In practice, the technique is most applicable to compounds which reach equilibrium rapidly. Otherwise, it takes a long time to achieve maximum effect at low toxicant levels. Here, too, application factors are needed to use data from bioassayed concentrations in estimating levels for environmental protection. Theoretically, application factors account for variations in sensitivity between the life stage tested and that life stage or developmental period during which the organism is most sensitive to the compound or conditions. Application factors should also safely permit a range of naturally-occurring environmental variations that would increase sensitivity.

Long-term bioassay, in which the organism is kept through at least one complete life cycle under conditions of continuous-flow exposure, is perhaps the closest but most conservative laboratory approach to estimating environmental hazards. Where a chemical or physical attraction occurs or where the population is sessile or restricted by hydrographic features, continuous exposure to freshly added material will be a realistic model. However, where the organism might escape in nature, such a captive exposure will be unrealistic. The experimental conditions chosen may either be held constant or varied to approximate local natural changes or intermittent discharges to be expected. Adequate modelling of a particular environmental circumstance often requires varying degrees of delay between the time of test material addition and exposure of the organisms.

Duration of chronic toxicity studies is determined by the life span and reproductive cycle of the organism chosen. Micro-organisms have relatively short life cycles but may require several generations to deplete metabolite reserves and show maximum response. A greater variety of measurements can be used in long-term than in short-term testing. This variety, together with the longer period available for response and the certainty of testing the most sensitive life stage, serves to increase both the sensitivity and relevance of such tests. Differences in sensitivity between species, that may be evident in short-term tests, tend to narrow as the tests approach a full life cycle.

The maintenance of a resident population of sensitive organisms in an effluent stream or portion of a natural stream receiving effluent, can create a long-term flow-through bioassay. This technique is primarily useful as a verification of safety based on other estimates, but because the response time may be long, the results are of little use where rapid feedback of information is essential.

DESIGN OF BIOASSAYS

The bioassay system may be compartmentalized for purposes of design into (1) the substance to be tested, (2) the environment into which it will be introduced, (3) the organism(s) which will be exposed to the resultant system, and (4) the observations to be made. Each affects and is affected by the others.

The chemical and physical nature of the material to be tested has a bearing on the way it will distribute in nature and in the test system and thus on which organisms will encounter it and in what form it will be. For example, a pure substance, highly soluble in water, may be tested for its effect directly on organisms inhabiting the water column. A material which precipitates rapidly may be readily available to organisms which ingest the precipitate and resolubilize it under conditions prevailing in the digestive tract. Materials which are only slightly soluble are often readily available to micro-organisms which have a high surface-area-to-volume ratio and are capable of taking up some substances at exceedingly low (10^{-8} to 10^{-10} M) concentrations. A highly hydrophobic material which is readily adsorbed to sediments or detritus may appear in free solution to only a limited extent or for a short time and exert a prolonged direct effect mainly on those organisms which inhabit sediments or which process sediments or detritus for food. Valid interpretation of bioassay results requires sensitive and highly specific analytical chemistry as part of the procedure. Results obtained for any bioassay organism are subject to question if anomalous behavior of the substance tested or the organisms used are subsequently established.

The organism for bioassay should be chosen on the basis of the relationship of its life stages to the various toxicant compartments and information desired. Organisms will be useful if they are readily available and can be reared and propagated in the laboratory. The size of the organisms in relation to available facilities will in part dictate a choice. All too often, these have been the primary if not the only considerations. There is a temptation to give priority to organisms that are available from standard sources with a known genetic line or from a single clone. This approach is essential when using bioassay as an analytical tool. However, it is a distinct liability when performing measurements of biological effect in natural environmental situations. Such organisms have necessarily undergone selection for traits that favor survival in artificial environments with no selective advantage given to the capacity to adapt to alterations in those environments. Furthermore, physiologically distinct races often develop in nature in response to characteristics of different localities. Maintenance of laboratory stocks may be necessary, but these stocks should be frequently renewed from fresh isolates representing the gene pool and enzymatic adaptations of the inhabitants of the particular water mass to which recommendations are to be applied.

The organisms used should be drawn from those that are most sensitive or respond most quickly to the substance or condition being tested. Bioassays of various life stages of these sensitive organisms are desirable. It is especially important that life stages to be tested include those that will most probably encounter the test material as it is expected to be found in the environment, and that the test organisms be acclimated to the test system until the characteristics to be measured become constant.

Some of the foregoing recommendations for selection assume that the developmental biology of the test organism is known. This is not often so in marine biology. Organisms should not be excluded from consideration if their absence would leave no representatives of local species which tolerate the extremes in ranges of natural environmental stress or which fill an important ecological niche.

Once an understanding of both the test material and the bioassay organism is established, a test system usually can be designed that will permit the organism to encounter the test material under circumstances approximating those in nature. In some cases it will be necessary to go to the natural water system or to impoundments, live cars, or plastic bags in order to obtain a workable approximation of environmental exposure. Care should be taken that the physical system does not interfere with the distribution of the test material or the behavior of the organism. The system selected should reflect in all important aspects the habitat to which the test organism has become adapted. Factors of importance include feeding behavior, opportunity for diurnal behavior alterations, emergence, salinity variations, turbidity, water movement, and other factors, depending on the organisms being studied.

The response or responses to be observed during long-term testing must be carefully chosen. A prime requirement is that the response being measured bear a demonstrable and preferably quantitative relationship to the survival and productivity of the test organism or of an organism which is directly or indirectly dependent on its activities. For example, a correlation may exist between the level of a test material and the amount of an enzyme present in some tissue. This is clear evidence that the organism's pattern of energy utilization has changed, but it should be demonstrated that the change in enzyme level is correlated with or predictive of changes in growth, behavior, reproduction, quality of flesh, or some other manifestation to provide an immediately meaningful interpretation.

The degree to which a response can be reported in quantitative terms affects its usefulness. Behavior, because of a high degree of variability, is much more difficult to express numerically than growth; and growth measurements are usually disruptive of the system or destructive of the organism. A balance must be sought for each system so that enough organisms and replicate treatments can be used to assure an acceptable level of statistical confidence in the results. Considerations of equipment required, rapidity, and simplicity of measurements, the inherent (control) variability of the characteristic being measured, and possible interference with the measurement by the substance being tested must enter into the choice of measurements and their frequency.

Biological characteristics that can be measured are innumerable, but some may be singled out as being more basic than others. When a given characteristic reflects many diverse processes, it is most useful in interpreting results in terms of environmental protection. Thus, measurements of reproductive success, growth, life span, adaptation to environmental stress, feeding behavior, morphology, respiration, histology, genetic alterations, and biochemical anomalies occupy a descending scale in order of the confidence that can be placed in their interpretation. This is not to say that profound changes in the structure and function of an ecosystem cannot result from subtle, prolonged, low intensity effects on some cellular process. The elimination of important species by low intensity selective factors is no less serious than instantaneous death of those species. In a sense, it is more serious, because it is less likely to be noticed and traced to its source in time to permit recovery of the ecosystem.

SUBLETHAL EFFECTS

Many biological effects of pollution may not show up in the bioassay test for acute toxicity. This would be true if the effect were slow to develop, or if the effect were to produce a general debility that might interfere with some of the normal life functions of the organism rather than killing it directly. Long-term exposure to sublethal concentrations may be necessary to produce the effect, and evaluation of this type of action is difficult in a laboratory analysis. There are a number of ways in which pollutants might affect a given population without being lethal to the adult organism used in the test such as:

Migrations

Sublethal concentrations may interfere with the normal migration patterns of organisms. The mechanisms used for orientation and navigation by migrating organisms are not well known, but in some cases chemotaxis clearly plays an important role. For example, salmon and many other anadromous fishes have been excluded from their home streams by pollution, though it is not known whether the reason is that a chemical cue has been masked or because the general chemical environment of pollution is offensive to the fish.

Behavior

Much of the day-to-day behavior of species may also be mediated by means of chemotaxic responses. The finding and capture of food or the finding of a mate during the breeding season would be included in this category of activity. Again, any pollutant that interfered with the chemoreceptors of the organism would interfere with behavioral patterns essential to the survival of the population.

Incidence of Disease

Long-term exposure to sublethal concentrations of pollutants may make an organism more susceptible to a disease. It is also possible that some pollutants which are organic in nature may provide an environment suitable for the development of disease-producing bacteria or viruses. In such cases, even though the pollutant is not directly toxic to the adult organism, it could have a profound effect on the population of the species over a longer period of time.

Life Cycle

The larval forms of many species of organisms are much more sensitive to pollution than are the adults, which are commonly used in the bioassay. In many aquatic species millions of eggs are produced and fertilized, but only two of the larvae produced need to grow to maturity and breed in order to maintain the standing stock of the species. For these species the pre-adult mortality is enormous even under the best of natural conditions. Because of an additional stress on the developing organisms, enough individuals might fail to survive to maintain the population of the species. Interrupting any stage of the life cycle can be as disastrous for the population as would death of the adults because of acute toxicity.

Physiological Processes

Interference with various physiological processes, without necessarily causing death in a bioassay test, may also interfere with the survival of the species. If photosynthesis of the phytoplankton is inhibited, algal growth will be decreased, and the population may be grazed to extinction without being directly killed by the toxin.

Respiration or various other enzymatic processes might also be adversely affected by sublethal concentrations of pollutants. The effect of DDT and its decomposition products on the shells of bird eggs is probably the result of interference with enzyme systems (Ackefors et al. 1970).[88] Mercury is a general protoplasmic poison, but it has its most damaging effect on the nervous system of mammals.

Genetic Effects

Many pollutants produce genetic effects that can have long range significance for the survival of a species. Oil and other organic pollutants may include both mutagenic and carcinogenic compounds. Radioactive contamination can cause mutations directly by the action of the radiation on the genetic material. From genetic studies in general, it is known that a large majority of mutations are detrimental to the survival of the young, and many are lethal. Little is known about the intensity or frequency of genetic effects of pollutants, except for radioactive materials where the mutation rates have been measured in some cases. Induction of mutation by contaminants should be reviewed in the context of the increase of total mutation from all causes.

Nutrition and Food Chains

Pollutants may interfere with the nutrition of organisms by affecting the ability of an organism to find its prey, by interfering with digestion or assimilation of food, or by contaminating the prey species so that it is not accepted by the predator. On the other hand, if predator species are eliminated by pollution, the prey species may have an improved chance of survival. An example of the latter effect was shown for the kelp resurgence after the oil spill in Tampico Bay, California (North 1967).[93] The oil killed the sea urchins which used young, newly developing kelp as food. When the urchins were killed, the kelp beds developed luxurious growth within a few months (see p. 258).

Effects on the Ecosystem

The effects of pollution on the aquatic ecosystem are the most difficult to evaluate and establish. Each environment is somewhat different, but the species inhabiting any given environment have evolved over long periods of time, and each individual species in a community plays its own role. Any additional stress, whether natural or man-made, applied to any environment will tend to eliminate some species leaving only the more tolerant forms to survive. The effect may be either direct on the species involved or indirect through the elimination of some species valuable as a food supply. For some of the species in the system the result may be beneficial by the removal of their predators or by stimulated and accelerated growth of their prey.

Food Value for Human Use

Sublethal concentrations of pollutants can so taint seafood that it becomes useless as a source of food. Oil can be ingested by marine organisms, pass through the wall of the gut, and accumulate in the lipid pool. Blumer (1971)[89] stated that oil in the tissues of shellfish has been shown to persist for months after an oil spill; the oil-polluted area was closed for shellfishing for a period of 18 months. Seafood may be rendered unfit for human consumption because of the accumulation of pollutants. California mackeral and coho salmon from Lake Michigan were condemned because they contained more DDT than the permissible amount in human food (5 mg/kg). Likewise tuna fish and swordfish were removed from the market, because the mercury content of the flesh exceeded the allowable concentration (0.5 mg/kg). There was no evidence that these concentrations had any adverse effect on the fish, or in the case of mercury that the concentrations in tuna and swordfish resulted from pollution; nevertheless their removal from the market has adversely affected the economics of the fisheries.

CATEGORIES OF POLLUTANTS

TEMPERATURE AND HEAT

An extensive discussion of heat and temperature is presented in Section III on Freshwater Aquatic Life and Wildlife (pp. 151–171). Although we accept those recommendations concerning temperature, there are certain characteristics of the marine environment that are unique and require enumeration. Some of the characteristics of the marine environment have been discussed in the introduction to this Section showing that the range of variability is greatest in the estuary, considerably less in the coastal waters, and even less in the surface waters of the open ocean; and that conditions in the deep ocean are virtually constant. Among the most important variables shown in the changes is temperature, although salinity variations are equally important under certain conditions.

The seasonal range of temperature variations is greatest in the temperate regions and becomes less as one approaches either the tropics or the poles. In the United States, the maximum seasonal temperature variation is found in the coastal waters on the southern side of Cape Cod, Massachusetts, where in winter the water may be freezing at −2.8 C and in summer the inshore coastal waters reach temperatures of 23 C, or even 25 C over wide shoal areas. At the same latitude on the Pacific coast, the water is neither so cold in the winter nor so warm in the summer. North of Cape Cod, the water is as cold in the winter time, but it does not reach as high a summer temperature; and south of Cape Cod the waters rarely reach a freezing point in winter.

Hutchins (1947)[100] discusses these ranges of variations and illustrates how they affect geographical distribution of marine species on the Atlantic European coasts and on the east and west coasts of the United States. As is obvious from the above comments, Cape Cod is a geographical boundary in the summertime but not in winter. Because temperature can control both the breeding cycle and survival of organisms, a variety of different geographical distributions can be dominated by the temperature variations at various locations along the coast (Hutchins 1947).[100]

There is increasing pressure to site power plants in the coastal zone because of the large available supply of water for cooling purposes. In 1969 there were over 86 fossil fuel power plants in the eastern coastal zones (Sorge 1969)[101] and 32 on the west coast (Adams 1969).[98] In addition, nuclear power plants are in operation, and many more are planned for siting on the coast in the future. Provided that the temperatures are kept within the limit prescribed in the recommendations and that the recommendations for mixing zones (pp. 228–232) are complied with, these heated effluents may have no serious impact on the marine environment. However, organisms passing through the cooling system of the power plants may be killed either by the direct effect of temperature, by pressure changes in the system, or by chlorination if it is used to keep the cooling system free of attached growth.

In the tropics, disposal of waste heat in the marine environment may be impossible in the summertime. Bader and Roessler (1972)[99] discussed the temperature problems created by the power plants at Turkey Point, near Miami, Florida. Thorhaug et al. (1972)[102] showed that tropical marine organisms live precariously close to their upper thermal limit and are thus susceptible to the stress of additional thermal effluents. To abide by the temperature recommendations in tropical waters, it is generally necessary to prohibit discharge of heated effluents during the summertime.

It is clear from this and from the discussion in Section III that additional studies will be needed on the temperature tolerances of the species directly involved. Organisms from estuaries and marine waters have not been studied as extensively as have freshwater fishes, but some data are included in the tabular material in the freshwater report. On the basis of information available at this time, the marine panel finds that the recommendations in Section III, Freshwater Aquatic Life and Wildlife, appear to be valid for the estuarine and marine waters as well (see pp. 160, 161, 164, 165, and 166–171 of Section III).

INORGANIC CHEMICALS, INCLUDING HEAVY METALS AND pH

The hazardous and biologically active inorganic chemicals are a source of both local and world-wide threats to the marine environment. Certain of these chemicals may pose no immediate danger but may lead to undesirable long-term changes. Others, such as boron, may pose serious health hazards and yet have poorly understood biological effects in the marine environment. Nevertheless, they can be a significant constituent in certain waste waters and should be discussed here.

The inorganic chemicals that have been considered in this study are listed alphabetically in Table IV-2; those most significant to the protection of the marine environment are discussed below.

TABLE IV-2—Inorganic Chemicals to be Considered in Water Quality Criteria for Aquatic Life in the Marine Environment

Elements	Equilibrium species (reaction)	Natural concentration in sea water[a] µg/l	Pollution categories[b]
Aluminum	$Al(OH)_3$, solubility of Al_2O_3 approx. 300 µg/l	10	IV c
Ammonia	NH_3, NH_4^+		IV c
Antimony	$Sb(OH)_6^-$	0.45	IV c ?
Arsenic	As_2O_3 is oxidized to $HAsO_4^{2-}$	2.6	II c
Barium	Ba^{2+}	20	IV c
Beryllium	$Be(OH)_2$, solubility of BeO approx. 10 µg/l	0.0006	IV c ?
Bismuth	$Bi(OH)_3$, solubility of Bi_2O_3 is unknown (low)	0.02	IV c ?
Boron	$B(OH)_3$, $B(OH)_4^-$	4.5×10^3	IV c
Bromine	Br^-, HBrO, Br^-	6.7×10^4	IV c
Cadmium	$CdCl^+$, $CdCl_2$, $CdCl_3^-$ (the last two are probably the main forms)	0.02	II c
Calcium	Ca^{2+}	4.1×10^5	IV c
Chlorine	Cl^-, HClO		IV c
Chromium	$Cr(OH)_3$, solubility of Cr_2O_3 unknown (low)	0.04	IV c ?
Cobalt	Co^{2+}	0.4	IV c
Copper	Cu^{2+}, $CuOH^+$, $CuHCO_3^+$, $CuCO_3$ (probably main form) $CuCl^+$, complexed also by dissolved amino acids	1	IV c
Cyanide	HCN (90%), CN^- (10%)		III c
Fluoride	F^- (50%), MgF^+ (50%)	1340	IV c
Gold	$AuCl_2^-$	.01-2	IV c
Hydrogen Ion (Acids)	$HCl + HCO_3^- \rightarrow H_2O + Cl^- + CO_2$ $H_2SO_4 + 2HCO_3^- \rightarrow 2H_2O + SO_4^{2-} + 2CO_2$	pH = 8 (alk = 0.0024 M)	III c
Iron	$Fe(OH)_3$, solubility of FeOOH approx. 5 µg/l	10	IV c
Lead	Pb^{2+}, $PbOH^+$, $PbHCO_3^+$, $PbCO_3$, $PbSO_4$, $PbCl^+$ (probably main form)	0.02	I a
Magnesium	Mg^{2+}	1.3×10^6	IV c
Manganese	Mn^{2+}	2	IV c
Mercury	$HgCl_2$, $HgCl_3^-$, $HgCl_4^{2-}$ (main form)	0.1	I b
Molybdenum	MoO_4^{2-}	10	IV c
Nickel	Ni^{2+}	7	III c
Nitrate	NO_3^-	6.7×10^2	III c
Phosphorus	Red phosphorus reacts slowly to phosphate $H_2PO_4^-$ and HPO_4^{2-}		
Selenium	SeO_4^{2-}	0.45	III c ?
Silicon	$Si(OH)_4$, $SiO(OH)_3^-$	3×10^3	IV c
Silver	$AgCl_2^-$	0.3	III c
Sulfide	S^{2-}		II c
Thallium	Tl^+	0.1	III c
Titanium	$Ti(OH)_4$, solubility of TiO_2 unknown (low)	2	IV b ?
Uranium	$UO_2(CO_3)_3^{4-}$	3	III c
Vanadium	VO_2OH	2	IV a ?
Zinc	Zn^{2+}, $ZnOH^+$, $ZnCO_3$, $ZnCl^+$ (probably main form)	2	III c

[a] These values are approximate but they are representative for low levels in unpolluted sea water.

[b] I-IV order of decreasing menace; a—worldwide, b—regional, c—local (coastal, bays estuaries, single dumpings). ? indicates some question of the ranking as a menace and/or whether the pollutional effect is local, regional, or worldwide.

Adapted and modified from the Report of the Seminar on Methods of Detection, Measurement and Monitoring of Pollutants in the Marine Environment. Food and Agriculture Organization 1971[164].

Forms of Chemical and Environmental Interactions

The form in which a chemical appears in the environment depends on the chemical and physical characteristics of the element, its stability, and the characteristics of the environment in which it is found. An element that is easily reduced or oxidized will undergo rapid changes, especially in sediments that alternate between oxidized and reduced states; while an element that is highly stable, such as gold, will retain its elemental identity in virtually all environmental conditions. Most elements are found in combined states, such as ore which can be a sulfide or a complex mineral containing oxygen, silica, and sulfur.

Certain elements are released into the environment by the processing of ores. Cadmium, for example, is not found uncombined in nature to any large extent but is a commercial by-product of zinc smelting. Other metallic elements can be brought into solution by the action of bacteria. Contamination from base metals may arise in abandoned mines, where tailings or slag heaps are attacked by physical and chemical weathering processes and bacteria to allow leaching of metallic ions into receiving waters. In strip mining, sulfides are oxidized to produce sulfuric acid, which may be a pollutant in itself or help to bring certain elements into solution.

The action of bacteria also transforms metals in another way. In anaerobic sediments, bacteria can convert inorganic metallic mercury into methyl mercury compounds. Such organo-metallic complexes are highly toxic to mammals, including man.

Biological Effects

Acute toxicity data for inorganic chemical compounds under controlled laboratory conditions, as represented for example by 96-hour LC50, are presented in Appendix III, Table 1, (pp. 449–460). Because of the lack of marine data, most of the information is based on freshwater bioassay data, which provide some measure of acute toxicity for the marine environment as well.

The concentrations of elements at which sublethal, chronic effects become manifest are also important. Sublethal concentrations of pollutants can have serious consequences in estuaries where migrating anadromous fishes linger to become acclimatized to changing salinities. Although the fish may not be killed outright, the stress of the sublethal concentrations may cause biochemical and physiological deficiencies that could impair life processes of the fish, preventing migrating adults from reaching their spawning grounds or reproducing. Pippy and Hare (1969)[247] suggested that heavy metals put fish under stress and may lead to infestation by diseases. Appendix III, Table 2 (pp. 461–468), summarizes data on the sublethal chronic effects of inorganic chemicals on fish and other aquatic organisms. As in Appendix III, Table 1, information on freshwater organisms has been included because of the paucity of tests in sea water. There is a clear need for toxicological work on the sublethal effects of pollutants on marine organisms.

At low concentrations, many elements are necessary to life processes, while at higher concentrations the same elements may be toxic. The effects of long-term exposure to low levels of most chemicals, singly or in combination, are generally unknown.

Laboratory bioassays are conducted under controlled conditions usually with single chemicals. Such tests provide toxicological information that must precede studies with mixtures closer to actual conditions. These mixtures must reflect the conditions and the composition of water in specific areas of discharge, because substances are rarely isolated when found in the environment. The probabilities of synergism and antagonism are enhanced by increased complexity of effluents. Synergism and antagonism in the environment are poorly understood. Copper is more toxic in soft water than in hard water where the calcium and the magnesium salts contributing to water hardness tend to limit or antagonize copper toxicity. Arsenic renders selenium less toxic and has been added to feeds for cattle and poultry in areas high in selenium. As examples of synergism, copper is considerably more toxic in the presence of mercury, zinc, or cadmium salts (LaRoche 1972),[211] and cadmium makes zinc and cyanide more toxic. Synergism or antagonism is expected to occur more frequently in water containing numerous chemical compounds than in one with few such compounds. Therefore, a complex chemical medium such as sea water can increase the probability of synergism or antagonism when a pollutant is introduced.

The effects of pollutants can be considered in terms of their biological end points. Such irreversible effects as carcinogenesis, mutagenesis, and teratogenesis provide identifiable end points in terms of biological consequences of pollutants. The effects of substances may vary with species or with stages of the life cycle (See Methods of Assessment, p. 233).

A distinction must be made between the effects of pollutants harmful to the quality of an organism as a product for human consumption and those harmful to the organism itself. While the levels of mercury that render fish unacceptable for marketing do not, on the basis of the limited information available at this time, appear to have any adverse effect on the fish themselves, they cause condemnation of the product for human consumption. This may also be true for other elements that lend themselves to bioaccumulation. Elemental phosphorus leads to illness and eventual mortality of fish themselves (Jangaard 1970).[191] At the concentrations of phosphorus found in the liver and other vital organs, the fish may have been toxic to human beings as well. The recommendations for the elements subject to biological accumulation in the marine environment must be set at a low level to protect the organisms. There is also need to establish recommendations based on human health, and a need to protect the economic value of fisheries affected by accumulations of some of these elements.

Data on the accumulation of inorganic chemicals by aquatic organisms are given in Appendix III, Table 3 (pp. 469–480). The maximum permissible concentrations of inorganic chemicals in food and water, as prescribed by the U.S. Food and Drug Administration and by drinking water standards of various agencies, are given in Appendix III, Table 4 (pp. 481–482).

The elements essential to plant and animal nutrition in the marine environments have been included in Table IV-2. They constitute some of the ordinary nutrients, e.g., silicon and nitrate, as well as the micro-constituents, such as iron, molybdenum, and cobalt. Although it is recognized that these elements are required for algal nutrition, one must not be caught in the misconception that "if a little is good, a lot is better."

Metals

Metals reach the marine environment through a variety of routes, including natural weathering as well as municipal and industrial discharges. Metals are particularly susceptible to concentration by invertebrates. Vinogradov's (1953)[294] classic work on the accumulation of metals by organisms in the marine environment has been expanded in more recent treatises (Fukai and Meinke 1962,[166] Polikarpov 1966,[249] Bowen et al. 1971,[129] Lowman et al. 1971).[221]

Metals present in the marine environment in an assimilable form usually undergo bioaccumulation through the food chain. Thus, elements present in low concentrations in the water may be accumulated many thousandfold in certain organisms. Established maximum permissible levels of some of these metallic ions render fish unacceptable for the commercial market (U.S. Department of Health, Education, and Welfare, Food and Drug Administration 1971,[286] Kolbye 1970[206]). Food and drug control agencies must impose stringent requirements on the content of certain hazardous elements, such as mercury, which, during 1970, led to condemnation of much of the fish caught in waters of the Canadian Prairies and the southern Great Lakes. Much of the swordfish and about 25 per cent of the tuna caught by the Japanese have exceeded the maximum permissible limit (Wallace et al. 1971).[295]

Studies conducted on Atlantic salmon (*Salmo salar*) in St. Andrew's, New Brunswick, show that low concentrations of zinc and copper mixtures will set up avoidance reactions (Sprague 1965,[268] Sprague and Saunders 1963[271]). Adult salmon migrating to spawn can be diverted by low concentrations of these base metals such as those leached from mine tailings. There are indications that as much as 25 per cent of spawning salmon (*Salmo salar*) may return to sea without going through the spawning act if concentrations of zinc and copper are high enough to induce avoidance reactions (Sprague 1965).[268] There may be other similar important behavioral reactions stimulated by low concentrations of some of the metals.

In the following review of different inorganic constituents, the total amount of each element is considered in the discussion and recommendation, unless otherwise stated. Whereas some of the methods of analysis for constituents recommended for fresh water and waste water can also be used in marine environments, the interference from salt demands other specialized techniques for many elements (Strickland and Parsons 1968,[273] Food and Agriculture Organization 1971[164]).

Not only has the recent literature been reviewed in this examination of the properties and effects of inorganic constituents, but various bibliographic and other standard references have been liberally consulted (The Merck Index 1960,[228] McKee and Wolf 1963,[226] Wilber 1969,[299] NRC Committee on Oceanography 1971,[257] U.S. Department of the Interior Federal Water Pollution Control Administration 1968,[287] Canada Interdepartmental Committee on Water 1971[136]).

Alkalinity or Buffer Capacity, Carbon Dioxide, and pH

The chemistry of sea water differs from that of fresh water largely because of the presence of salts, the major constituents of which are present in sea water in constant proportion. The weak-acid salts, such as the carbonates, bicarbonates, and borates, contribute to the high buffering capacity or alkalinity of sea water. This buffering power renders many wastes of a highly acidic or alkaline nature, which are often highly toxic in fresh water, comparatively innocuous after mixing with sea water.

The complex carbon dioxide-bicarbonate-carbonate system in the sea is described in standard textbooks (Sverdrup et al. 1946,[276] Skirrow 1965[261]). Alkalinity and the hydrogen-ion concentration, as expressed by pH (Strickland and Parsons 1968),[273] are the best measure of the effects of highly acidic or highly alkaline wastes.

European Inland Fisheries Advisory Commission (1969)[158] and Kemp (1971)[212] reviewed the pH requirements of freshwater fishes. Because of the large difference in buffer capacities, techniques for measurement and definitions of alkalinity are quite different for marine and fresh waters. The normal range of pH encountered in fresh water is considerably wider than that found in sea water, and for this reason, freshwater communities are adapted to greater pH extremes than are marine communities.

Sea water normally varies in pH from surface to bottom because of the carbon dioxide-bicarbonate-carbonate equilibria. Photosynthetic and respiratory processes also contribute to variations in pH. At the sea surface, the pH normally varies from 8.0 to 8.3, depending on the partial pressure of carbon dioxide in the atmosphere and the salinity and temperature of the water. A large uptake of carbon dioxide during photosynthesis in the euphotic zone leads to high pH values exceeding 8.5 in exceptional cases. Release of carbon dioxide during decomposition in intermediate and bottom waters results in a lowering of pH. In shallow, biologically-active waters, particularly in warm tropical and subtropical areas, there is a large diurnal variation in pH with values ranging from a high of 9.5 in the daytime to a low of 7.3 at night or in the early morning.

The toxicity of most pollutants increases as the pH increases or decreases from neutral (pH 7). This is true for complex mixtures, such as pulp mill effluents (Howard and Walden 1965),[183] for constituents which dissociate at different pH (e.g., H_2S and HCN), and for heavy metals. The toxicity of certain complexes can change drastically with pH. Nickel cyanide exhibits a thousandfold increase in toxicity with a 1.5 unit decrease in pH from 8.0 to 6.5 (Robert A. Taft Sanitary Engineering Center 1953,[255] Doudoroff et al. 1966[152]). pH may also determine the degree of dissociation of salts, some of which are more toxic in the molecular form than in the ionic form. Sodium sulfide is increasingly toxic with decreasing pH as $S^{=}$ and HS^{-} ions are converted to H_2S (Jones 1948).[200] The tolerance of fish to low concentrations of dissolved oxygen, high temperatures, cations, and anions varies with pH. Therefore, non-injurious pH deviations and ranges depend on local conditions.

There are large fluctuations in natural pH in the marine environment. Changes in pH indicate that the buffering capacity of the sea water has been altered and the carbon dioxide equilibria have shifted. The time required for mixing of an effluent with a large volume of sea water is exceedingly important. When the pH of the receiving sea water undergoes an increase or decrease, its duration can be important to the survival of organisms. At present, there are not sufficient data with which to assign time limits to large departures of pH.

Fish tolerate moderately large pH changes in the middle of their normal pH ranges. Small pH changes at the limits of their ranges and also in the presence of some pollutants can have significant deleterious effects.

Plankton and benthic invertebrates are probably more sensitive than fish to changes in pH. Oysters appear to perform best in brackish waters when the pH is about 7.0. At a pH of 6.5 and lower, the rate of pumping decreases notably, and the time the shells remain open is reduced by 90 per cent (Loosanoff and Tommers 1948,[219] Korringa 1952[207]). Oyster larvae are impaired at a pH of 9.0 and killed at 9.1 in a few hours (Gaarder 1932).[167] The upper pH limit for crabs is 10.2 (Meinck et al. 1956).[227]

Recommendation

Changes in sea water pH should be avoided. The effects of pH alteration depend on the specific conditions. In any case, the normal range of pH in either direction should not be extended by more than 0.2 units. Within the normal range, the pH should not vary by more than 0.5 pH units. Ad-

dition of foreign material should not drop the pH below 6.5 or raise it above 8.5.

Aluminum

Aluminum, one of the most abundant elements in the earth's crust, does not occur in its elemental form in nature. It is found as a constituent in all soils, plants, and animal tissues. Aluminum is an amphoteric metal; it may be in solution as a weak acid, or it may assume the form of a flocculent hydroxide, depending on the pH. In the aluminum sulfate form (alum), it is used in water treatment as a coagulant for suspended solids, including colloidal materials and microorganisms.

Aluminum may be adsorbed on plant organisms, but very little ingested by animals is absorbed through the alimentary canal. Goldberg et al. (1971)[172] reported an aluminum concentration factor for phytoplankton (*Sargassum*) ash of 65 and for zooplankton ash of 300. However, Lowman et al. (1971),[221] in their compilation of concentration factors for various elements, noted that aluminum was reported to be concentrated 15,000 times in benthic algae, 10,000 times in plankton (phyto- and zoo-), 9,000 times in the soft parts of molluscs, 12,000 times in crustacean muscle, and 10,000 times in fish muscle.

In fresh water, the toxicity of aluminum salts varies with hardness, turbidity, and pH. Jones (1939)[198] found the lethal threshold of aluminum nitrate for stickleback (*Gasterosteus aculeatus*) in very soft water to be 0.07 mg/l. Using tap water with the same compound tested on the same species, Anderson (1948)[112] reported a toxic threshold of less than 5×10^{-5} molar aluminum chloride (1.35 mg/l Al). Average survival times of stickleback in different concentrations of aluminum in the nitrate form have been reported as one day at 0.3 mg/l and one week at 0.1 mg/l (Doudoroff and Katz 1953).[150] It was noted by the same authors that 0.27 mg/l aluminum in the nitrate form did not apparently harm young eels in 50 hours' exposure.

Because of the slightly basic nature of sea water, aluminum salts tend to precipitate in the marine environment. These salts have exhibited comparatively low toxicities with 96-hour LC50's of 17.8 mg/l for redfish tested in sea water with aluminum chloride (Pulley 1950).[252] Concentrations of 8.9 mg/l of aluminum (from $AlCl_3$) did not have a lethal effect on marine fish and oysters tested (*Cynoscion nebulosus, Sciaenops oscellatus, Fundulus grandis, Fundulus similis, Cyprindon variegatus, Ostrea virginica*) (Pulley 1950).[252] The flocs of precipitated aluminum hydroxide may affect rooted aquatics and invertebrate benthos. Wilder (1952)[300] noted no significant effect on lobsters (*Homarus americanus*) of a tank lined with an aluminum alloy (Mn, 1 to 1.5 per cent; Fe, 0.7 per cent; Si, 0.6 per cent; Cu, 0.2 per cent, and Zn, 0.1 per cent).

Aluminum hydroxide can have an adverse effect on bottom communities. Special precautions should be taken to avoid disposal of aluminum-containing wastes in water supporting commercial populations of clams, scallops, oysters, shrimps, lobsters, crabs, or bottom fishes.

Recommendation

Because aluminum tends to be concentrated by marine organisms, it is recommended that an application factor of 0.01 be applied to marine 96-hour LC50 data for the appropriate organisms most sensitive to aluminum. On the basis of data available at this time, it is suggested that concentrations of aluminum exceeding 1.5 mg/l constitute a hazard in the marine environment, and levels less than 0.2 mg/l present minimal risk of deleterious effects.

Ammonia

Most of the available information on toxicity of ammonia is for freshwater organisms. For this reason, the reader is referred to the discussion of ammonia in Section III on Freshwater Aquatic Life and Wildlife (p. 186). Because of the slightly higher alkalinity of sea water and the larger proportion of un-ionized ammonium hydroxide, ammonia may be more toxic in sea water than in fresh water (Doudoroff and Katz 1961).[151] Holland et al. (1960)[182] noted a reduction in growth and a loss of equilibrium in chinook salmon (*Oncorhynchus tshawytscha*) at concentrations 3.5 to 10 mg/l of ammonia. Dissolved oxygen and carbon dioxide decrease the toxicity of ammonia (U.K. Department of Science and Research 1961).[284] Lloyd and Orr (1969),[217] in their studies on the effect of un-ionized ammonia at a pH of 8 to 10, found 100 per cent mortality with 0.44 mg/l NH_3 in 3 hours for rainbow trout (*Salmo gairdneri*). This confirmed earlier results of 100 per cent mortality in 24 hours at 0.4 mg/l. The toxicity increased with pH between 7.0 and 8.2.

Recommendation

It is recommended that an application factor of 0.1 be applied to marine 96-hour LC50 data for the appropriate organisms most sensitive to ammonia. On the basis of freshwater data available at this time, it is suggested that concentrations of un-ionized ammonia equal to or exceeding 0.4 mg/l constitute a hazard to the marine biota, and levels less than 0.01 mg/l present minimal risk of deleterious effects.

Antimony

Antimony occurs chiefly as sulfide (stibnite) or as the oxides cervantite (Sb_2O_4) and valentinite (Sb_2O_3) and is used for alloys and other metallurgical purposes. It has also been used in a variety of medicinal preparations and in numerous industrial applications. Antimony salts are used in the fireworks, rubber, textile, ceramic, glass and paint industries.

Few of the salts of antimony have been tested on fish in bioassays, particularly in sea water. However, antimony potassium tartrate ("tartar emetic") gave a 96-hour LC50 as antimony of 20 mg/l in soft water and 12 mg/l in hard water (Tarzwell and Henderson 1956,[277] 1960[278]). Cellular division of green algae was hindered at 3.5 mg/l, and movement of *Daphnia* was retarded at 9 mg/l (Bringmann and Kuhn 1959a).[131] Antimony trichloride, used in acid solution as a mordant for patent leather and in dyeing, was examined in exploratory tests on fathead minnows (*Pimephales promelas*) and gave a 96-hour LC50 as antimony of 9 mg/l in soft water and 17 mg/l in hard water (Tarzwell and Henderson 1960).[278] Applegate et al. (1957)[114] reported that rainbow trout (*Salmo gairdneri*), bluegill sunfish (*Lepomis macrochirus*), and sea lamprey (*Pertomyzon marinus*) were unaffected by 5 mg/l of $SbCl_3$ or $SbCl_5$ in Lake Huron water at 13 C, saturated with dissolved oxygen, and pH 7.5 to 8.2. Jernejcic (1969)[193] noted that as little as 1.0 mg/l of antimony in the form of tartar emetic caused projectile vomiting in large mouth bass (*Micropterus salmoides*).

Antimony can be concentrated by various marine forms to over 300 times the amount present in sea water (Goldberg 1957,[171] Noddack and Noddack 1939[240]).

Recommendation

Because of the hazard of antimony poisoning to humans and the possible concentration of antimony by edible marine organisms, it is recommended that an application factor of 0.02 be applied to marine 96-hour LC50 data for the appropriate organisms most sensitive to antimony. On the basis of data available at this time, it is suggested that concentrations of antimony equal to or exceeding 0.2 mg/l constitute a hazard in the marine environment. There are insufficient data available at this time to recommend a level that would present minimal risk of deleterious effects.

Arsenic

Arsenic occurs in nature mostly as arsenides or pyrites. It is also found occasionally in the elemental form. Its consumption in the U.S. in 1968 amounted to 25,000 tons as AS_2O_3 (U.S. Department of the Interior, Bureau of Mines 1969).[289] Arsenic is used in the manufacture of glass, pigments, textiles, paper, metal adhesives, ceramics, linoleum, and mirrors (Sullivan 1969),[274] and its compounds are used in pesticides, wood preservatives, paints, and electrical semiconductors. Because of its poisonous action on microorganisms and lower forms of destructive aquatic organisms, it has been used in wood preservatives, paints, insecticides, and herbicides. Sodium arsentite has been used for weed control in lakes and in electrical semiconductors.

In small concentrations, arsenic is found naturally in some bodies of water. In its different forms, including its valence states, arsenic varies in toxicity. Trivalent arsenic is considerably more toxic than the pentavalent species in the inorganic form. It is acutely toxic to invertebrates and for this reason has found application in the control of *Teredo* and other woodborers in the AS^{+3} form. Arsenious trioxide (As_2O_3) has been used for control of the shipworm *Bankia setacia*. In the arsenate form (As^{+5}), it is of relatively low toxicity, *Daphnia* being just immobilized at 18 to 31 mg/l sodium arsenate, or 4.3 to 7.5 mg/l as arsenic, in Lake Erie water (Anderson 1944,[110] 1946[111]). The lethal threshold of sodium arsenate for minnows has been reported as 234 mg/l as arsenic at 16 to 20 C (Wilber 1969).[299]

Arsenic is normally present in sea water at concentrations of 2 to 3 μg/l and tends to be accumulated by oysters and other molluscan shellfish (Sautet et al. 1964,[258] Lowman et al. 1971[221]). Wilber (1969)[299] reported concentrations of 100 mg/kg in shellfish. Arsenic is a cumulative poison and has long-term chronic effects on both aquatic organisms and on mammalian species. A succession of small doses may add up to a final lethal dose (Buchanan 1962).[135] The acute effects of arsenic and its compounds on aquatic organisms have been investigated, but little has been done on the sublethal chronic effects.

Surber and Meehan (1931)[275] found that fish-food organisms generally can withstand concentrations of approximately 1.73 mg/l of arsenious trioxide in sodium arsenite solution. Meinck et al. (1956)[227] reported that arsenic concentrations were toxic at 1.1 to 2.2 mg/l to pike perch (*Stizostedion vitreum*) in 2 days, 2.2 mg/l to bleak in 3 days, 3.1 mg/l to carp (*Cyrinus carpio*) in 4 to 6 days and to eels in 3 days, and 4.3 mg/l to crabs in 11 days.

Recommendation

Because of the tendency of arsenic to be concentrated by aquatic organisms, it is recommended that an application factor of 0.01 be applied to marine 96-hour LC50 data for the appropriate organisms most sensitive to arsenic. On the basis of freshwater and marine toxicity data available, it is suggested that concentrations of arsenic equal to or exceeding 0.05 mg/l constitute a hazard in the marine environment, and levels less than 0.01 mg/l present minimal risk of deleterious effects.

Barium

Barium comes largely from ores ($BaSO_4$, $BaCO_3$). It is being used increasingly in industry. The U.S. consumption in 1968 was 1.6 million tons, a growth of 78 per cent in 20 years (U.S. Department of the Interior, Bureau of Mines 1969).[289] Barium is used in a variety of industrial applications, including paper manufacturing, fabric printing and dyeing, and synthetic rubber production.

All water- or acid-soluble barium compounds are poisonous. However, in sea water the sulfate and carbonate present tend to precipitate barium. The concentration of barium in sea water is generally accepted at about 20 μg/l

(Goldberg et al. 1971),[172] although it has been reported as low as 6.2 μg/l (Bowen 1956).[128] Wolgemuth and Broecker (1970)[303] reported a range of 8 to 14 μg/l in the Atlantic and 8 to 31 μg/l in the Pacific, with the lower values in surface waters. Barium ions are thought to be rapidly precipitated or removed from solution by adsorption and sedimentation.

Bijan and Deschiens (1956)[123] reported that 10 to 15 mg/l of barium chloride were lethal to an aquatic plant and two species of snails. Bioassays with barium chloride showed that a 72-hour exposure to 50 mg/l harmed the nervous system of coho salmon (*Oncorhynchus kisutch*) and 158 mg/l killed 90 per cent of the test species (ORSANCO 1960).[245] Barium can be concentrated in goldfish (*Carassius auratus*) by a factor of 150 (Templeton 1958).[279] Soviet marine radioactivity studies showed accumulation of radioactive barium in organs, bones, scales, and gills of fish from the Northeast Pacific (Moiseev and Kardashev 1964[230]). Lowman et al. (1971)[221] listed a concentration factor for barium of 17,000 in phytoplankton, 900 in zooplankton, and 8 in fish muscle.

In view of the widespread use of barium, the effects of low doses of this element and its compounds on marine organisms under different environmental conditions should be determined. Disposal of barium-containing wastes into waters when precipitates could affect rooted aquatics and benthic invertebrates should be avoided.

Recommendation

Because of the apparent concentration of barium by aquatic organisms and the resultant human health hazard, it is recommended that an application factor of 0.05 be applied to marine 96-hour LC50 data for the appropriate organisms most sensitive to barium. On the basis of data available at this time, it is suggested that concentrations of barium equal to or exceeding 1.0 mg/l constitute a hazard in the marine environment, and levels less than 0.5 mg/l present minimal risk of deleterious effects.

Beryllium

Beryllium is found mainly in the mineral beryl and is almost nonexistent in natural waters. Its concentration in sea water is 6×10^{-4} μg/l. Beryllium is used in a number of manufacturing processes, in electroplating, and as a catalyst in organic chemical manufacture. It has also been used experimentally in rocket fuels and in nuclear reactors (Council on Environmental Quality 1971).[144] In 1968, the U.S. consumption of beryllium was 8,719 tons, a 500 per cent increase over 1948 (U.S. Department of the Interior, Bureau of Mines 1969).[289]

Beryllium has been shown to inhibit photosynthesis in terrestrial plants (Bollard and Butler 1966).[127] It would be of interest to know if there is any inhibition of photosynthesis by beryllium compounds in the marine environment.

Beryllium chloride and nitrate are highly soluble in water, and the sulfate is moderately so. The carbonate and hydroxide are almost insoluble in cold water. Toxicity tests gave a 96-hour LC50 for beryllium chloride of 0.15 mg/l as beryllium for fathead minnows (*Pimephales promelas*) in soft water; 15 mg/l for the same species in hard water (Tarzwell and Henderson 1960);[278] and 31.0 mg/l for *Fundulus heteroclitus* (Jackim et al. 1970).[190]

Beryllium has been reported to be concentrated 1000 times in marine plants and animals (Goldberg et al. 1971).[172]

Recommendation

In the absence of data specifically related to effects of beryllium on marine organisms, and because of its accumulation by marine organisms and its apparent toxicity to humans, it is recommended that an application factor of 0.01 be applied to marine 96-hour LC50 data for the appropriate organisms most sensitive to beryllium. On the basis of data available for hard fresh water, it is suggested that concentrations of beryllium equal to or exceeding 1.5 mg/l constitute a hazard to marine organisms, and levels less than 0.1 mg/l present minimal risk of deleterious effects.

Bismuth

Bismuth is used in the manufacture of bismuth salts, fusible alloys, electrical fuses, low-melting solders, and fusible boiler plugs, and in tempering baths for steel, in "silvering" mirrors, and in dental work. Bismuth salts are used in analytical chemical laboratories and commonly formulated in pharmaceuticals.

The concentration of bismuth in sea water is low, about 0.02 μg/l, probably because of the insolubility of its salts. It is unknown how much bismuth actually gets into the sea from man-made sources, but the quantity is probably small. The total U.S. production in 1969 as subcarbonate $(Bi_2O_2CO_3)_2 \cdot H_2O$ was 57 short tons (U.S. Department of Commerce 1971).[285]

There are no bioassay data on which to base recommendations for bismuth in the marine environment.

Boron

Boron is not found in its elemental form in nature; it normally occurs in mineral deposits as sodium borate (borax) or calcium borate (colemanite). The concentration of boron in sea water is 4.5 mg/l as one of the 8 major constituents in the form of borate. Boron has long been used in metallurgy to harden other metals. It is now being used in the elemental form as a neutron absorber in nuclear installations.

Available data on toxicity of boron to aquatic organisms are from fresh water (Wurtz 1945,[306] Turnbull et al. 1954,[281] LeClerc and Devlaminck 1955,[214] Wallen et al. 1957,[296] LeClerc 1960[213]). Boric acid at a concentration of 2000 mg/l showed no effect on one trout and one rudd (*Scardinius erythrophthalmus*); at 5000 mg/l it caused a discoloration of the skin of the trout, and at 80,000 mg/l the trout became immobile and lost its balance in a few minutes (Wurtz 1945).[306] The minimum lethal dose for minnows exposed to boric acid at 20 C for 6 hours was reported to be 18,000 to 19,000 mg/l in distilled water and 19,000 to 19,500 mg/l in hard water (LeClerc and Devlaminck 1955,[214] LeClerc 1960[213]). Testing mosquito fish (*Gambusia affinis*) at 20 to 26 C and a pH range of 5.4 to 9.1, Wallen et al. (1957)[296] established 96-hour LC50's of 5,600 mg/l for boric acid and 3,600 mg/l for sodium borate.

Since the toxicity is slightly lower in hard water than in distilled water, it is anticipated that boric acid and borates would be less toxic to marine aquatic life than to freshwater organisms. In the absence of sea water bioassay data, an estimate of 500 mg/l of boron as boric acid and [illegible] mg/l as sodium borate is considered hazardous to marine animals, based on freshwater data (Wallen et al. 1957).[296] Concentrations of 50 mg/l and 25 mg/l, respectively, are expected to have minimal effects on marine fauna.

An uncertainty exists concerning the effect of boron on marine vegetation. In view of harm that can be caused to terrestrial plants by boron in excess of 1 mg/l (Wilber 1969),[299] special precautions should be taken to maintain boron at normal levels near eel grass (*Zostera*), kelp (*Macrocystis*), and other seaweed beds to minimize damage to these plants.

Recommendation

On the basis of data available at this time, it is suggested that concentrations of boron equal to or exceeding 5.0 mg/l constitute a hazard in the marine environment, and levels less than 5.0 mg/l present minimal risk of deleterious effects. An application factor of 0.1 is recommended for boron compounds applied to marine 96-hour LC50 data for the appropriate organisms most sensitive to boron.

Bromine

In concentrated form, bromine is a strong oxidizing agent and will attack all metals and organic materials. It is one of the major constituents in sea water, present at about 67 mg/l in bromate, and is commercially extracted from the sea.

Bromine is used medicinally and for sterilization of swimming pools. It is also used in the preparation of dyestuffs and anti-knock compounds for gasolines. Molecular bromine may be discharged in effluents from salt works and certain chemical industries. Bromination of certain organic substances, such as phenols and amines, may impart offensive taste and make waters more toxic to aquatic organisms.

Kott et al. (1966)[208] found that *Chlorella pyrenoidosa*, when exposed to 0.42 mg/l bromine for 4 days, were reduced in concentration from 2,383 cells/mm² to 270 cells, but remained virtually unchanged at 0.18 mg/l bromine (2,383 cells/mm² in controls compared to 2,100 cells/mm² in the exposed sample).

At concentrations of 10 mg/l in soft water, bromine killed *Daphnia magna* (Ellis 1937),[156] and at 20 mg/l in water of 18 to 23 C, goldfish (*Carassius auratus*) were killed (Jones 1957).[201] A violent irritant response in marine fish was observed at 10 mg/l bromine, but no such activity was perceived at 1 mg/l (Hiatt et al. 1953).[181]

The salts of bromine are relatively innocuous. The threshold of immobilization for *Daphnia magna* was 210 mg/l of sodium bromate ($NaBrO_3$) and 8200 mg/l of sodium bromide (NaBr) (Anderson 1946).[111]

Recommendation

It is recommended that free (molecular) bromine in the marine environment not exceed 0.1 mg/l and that ionic bromine in the form of bromate be maintained below 100 mg/l.

Cadmium

U.S. consumption of cadmium was 6,662 short tons in 1968 (U.S. Department of the Interior, Bureau of Mines 1969).[289] These quantities indicate that cadmium might be a significant pollutant.

Pure cadmium is not found in commercial quantities in nature. It is obtained as a by-product of smelting zinc. Cadmium salts in high concentrations have been found in a Missouri spring originating from a mine (up to 1,000 mg/ml cadmium) (ORSANCO 1955),[244] and up to 50 to 170 mg/kg of cadmium are found in superphosphate fertilizers (Athanassiadis 1969).[116] Cadmium is also present in some pesticides. It is being used in increasing amounts by industry (Council on Environmental Quality 1971).[144] Water-carrying pipes are also a source of cadmium (Schroeder 1970)[259] as is food (Nilsson 1969).[239] Cadmium is present in most drainage waters (Kroner and Kopp 1965)[209] and may be contributing substantially to the cadmium present in inshore coastal waters. It is not known, however, whether man's input has resulted in higher levels of cadmium in estuarine or coastal waters. In sea water, cadmium is generally present at about 0.1 µg/l (Goldberg et al. 1971).[172]

Cadmium pollution resulting in the "Itai-itai" disease in the human population has been documented (Yamagata and Shigematsu 1970).[307] Schroeder et al. (1967)[260] have found that oysters may concentrate cadmium from very low levels in ambient water. Cadmium concentrations in some marine plants and animals have been given by Mullin and Riley (1956).[233]

Concern exists that cadmium may enter the diet, like mercury, through seafood. Cadmium, like mercury, could conceivably form organic compounds which might be highly toxic or lead to mutagenic or teratogenic effects.

Cadmium has marked acute and chronic effects on aquatic organisms. It also acts synergistically with other metals. A 15-week LC50 of 0.1 mg/l and inhibition of shell growth for *Crassostrea virginica* (Pringle et al. 1968),[250] and a 96-hour LC50 of 0.03 mg/l cadmium in combination with 0.15 mg/l zinc for fry of chinook salmon (*Oncorhynchus tshawytscha*) (Hublou et al. 1954)[184] have been reported.

Fundulus heteroclitus exposed to 50 mg/l cadmium showed pathological changes in the intestinal tract after 1-hour exposure, in the kidney after 12 hours, and in the gill filaments and respiratory lamellae after 20 hours (Gardner and Yevich 1970).[170] Copper and zinc, when present at 1 mg/l or more, substantially increase the toxicity of cadmium (LaRoche 1972).[211]

Cadmium is concentrated by marine organisms, particularly the molluscs (e.g., *Pecten novazetlandicae*), which accumulated cadmium in the calcareous tissues and in the viscera (Brooks and Rumbsby 1965).[133] Lowman et al. (1971)[221] reported a concentration factor of 1000 for cadmium in fish muscle.

Cadmium levels in tissues of Ashy Petrel (*Oceanodroama homochroa*) from coastal waters of California were approximately twice as high as in tissues of Wilson's Petrel (*Oceanites oceanicus*) obtained in Antarctica, which had summered in the North Atlantic and Australian regions, respectively. Cadmium levels in tissues of the Snow Petrel (*Pelagodroma nivea*), a species which does not leave the Antarctic ice pack region, obtained at Hallett Station, Antarctica, were of the same order of magnitude as those in the Wilson's Petrel. Cadmium levels in eggs of the Common Tern (*Sterna hirundo*) from Long Island Sound were in the order of 0.2 mg/kg dry weight, not appreciably higher than those in the Antarctic Tern (*Sterna vittata*) from the Antarctic with levels in the order of 0.1 mg/kg (Anderlini et al. *in press*).[109] Cadmium pollution may therefore be significant locally in estuaries, but on the basis of these limited data, it does not appear to be a problem in more remote marine ecosystems. However, in view of the comparatively unknown effects of cadmium on the marine ecosystem, its apparent concentration by marine organisms, and the human health risk involved in consumption of cadmium-contaminated seafood, it is suggested that there be no artificial additions of cadmium to the marine environment.

Recommendation

The panel recommends that an application factor of 0.01 be applied to marine 96-hour LC50 data for appropriate organisms most sensitive to cadmium. On the basis of data available at this time, it is suggested that concentrations of cadmium equal to or exceeding 0.01 mg/l constitute a hazard in the marine environment as well as to human populations, and levels less than 0.2 μg/l present minimal risk of deleterious effects. In the presence of copper and/or zinc at 1 mg/l or more, there is evidence that the application factor for cadmium should be lower by at least one order of magnitude.

In the absence of sufficient data on the effects of cadmium upon wildlife, it is recommended that cadmium criteria for aquatic life apply also to wildlife.

Chlorine

Chlorine is generally present in the stable chloride form which constitutes about 1.9 per cent of sea water. Elemental chlorine, which is a poisonous gas at normal temperature and pressure, is produced by electrolysis of a brine solution. Among its many uses are the bleaching of pulp, paper and textiles, and the manufacture of chemicals.

Chlorine is used to kill so-called nuisance organisms that might interfere with the proper functioning of hydraulic systems. Chlorine disinfection is also used in public water supplies and in sewage effluents to insure that an acceptable degree of coliform reduction is achieved before the effluents enter various bodies of water. In all instances the intent is to eliminate undesirable levels of organisms that would degrade water uses. This goal is only partially reached, because the effect of chlorine on desirable species is a serious hazard.

When dissolved in water, chlorine completely hydrolizes to form hypochlorous acid (HOCl) or its dissociated ions; at concentrations below 1000 mg/l, no chlorine exists in solution as Cl_2. The dissociation of HOCl to H^+ and OCl^- depends on the pH: 4 per cent is dissociated at pH 6, 25 per cent at pH 7, and 97 per cent at pH 9. The undissociated form is the most toxic (Moore 1951).[231] Although free chlorine is toxic in itself to aquatic organisms, combinations of chlorine with ammonia, cyanide, and organic compounds, such as phenols and amines, may be even more toxic and can impart undesirable flavors to seafood.

Chlorine at 0.05 mg/l was the critical level for young Pacific salmon exposed for 23 days (Holland et al. 1960).[182] The lethal threshold for chinook salmon (*Oncorhynchus tshawytscha*) and coho salmon (*O. kisutch*) for 72-hour exposure was noted by these investigators to be less than 0.1 mg/l chlorine. In aerated freshwater, monochloramines were more toxic than chlorine and dichloramine more toxic than monochloramine. Studies of irritant responses of marine fishes to different chemicals (Hiatt et al. 1953)[181] showed a slight irritant activity at 1 mg/l and violent irritant activity at 10 mg/l. Oysters are sensitive to chlorine concentrations of 0.01 to 0.05 mg/l and react by reducing pumping activity. At Cl_2 concentrations of 1.0 mg/l effective pumping could not be maintained (Galtsoff 1946).[169]

Preliminary results show that at 15 C, salinity 30 parts per thousand (‰), mature copepods (*Acartia tonsa* and

TABLE IV-3—Copepod Mortality from Chlorine Exposure

Acartia Tonsa

Chlorine mg/l	Exposure time in minutes to give 50 percent mortality	Exposure time in minutes to give 100 percent mortality
1.0	220	>500
2.5	8.5	120
5.0	1.2	10.0
10.0	0.6	1.0

Eurytemona Affinis

Chlorine mg/l	Exposure time in minutes to give 50 percent mortality	Exposure time in minutes to give 100 percent mortality
2.5	33	125
5.0	3.6	30.0
10.0	2.0	5.0

Gentile (unpublished data) 1972.[312]

Eurytemona affinis) have great difficulty in surviving exposures to chlorine (Table IV-3).

Clendenning and North (1960)[141] noted that at 5 to 10 mg/l chlorine, the photosynthetic capacity of bottom fronds of the giant kelp (*Macrocystis pyrifera*) was reduced by 10 to 15 per cent after 2 days and 50 to 70 per cent after 5 to 7 days.

Chlorination in seawater conduits to a residual of 2.5 mg/l killed all fouling organisms tested (anemones, mussels, barnacles, Mogula, Bugula) in 5 to 8 days; but with 1.0 mg/l a few barnacles and all anemones survived 15 days' exposure (Turner et al. 1948).[282]

It should be further stressed that chlorine applications may often be accompanied by entrainments where the organisms are exposed to strong biocidal chlorine doses, intense turbulence, and heat (Gonzales et al. *unpublished* 1971).[313] Consideration should also be given to the formation of chlorinated products, such as chloramines or other pollutants, which may have far greater and more persistent toxicity than the original chlorine applications.

Recommendation

It is recommended that an application factor of 0.1 be used with 96-hour LC50 data from seawater bioassays for the most sensitive species to be protected.

However, it is suggested that free residual chlorine in sea water in excess of 0.01 mg/l can be hazardous to marine life. In the absence of data on the *in situ* production of toxic chlorinated products, it appears to be premature to advance recommendations.

Chromium

Most of the available information on toxicity of chromium is for freshwater organisms, and it is discussed in Section III, p. 180.

Chromium concentrations in seawater average about 0.04 µg/l (Food and Agriculture Organization 1971),[164] and concentration factors of 1,600 in benthic algae, 2,300 in phytoplankton, 1,900 in zooplankton, 440 in soft parts of molluscs, 100 in crustacean muscle, and 70 in fish muscle have been reported (Lowman et al. 1971).[221]

The toxicity of chromium to aquatic life will vary with valence state, form, pH, synergistic or antagonistic effects from other constituents, and the species of organism involved.

In long-term studies on the effects of heavy metals on oysters, Haydu (*unpublished data*)[314] showed that mortalities occur at concentrations of 10 to 12 µg/l chromium, with highest mortality during May, June, and July. Raymont and Shields (1964)[253] reported threshold toxicity levels of 5 mg/l chromium for small prawns (*Leander squilla*), 20 mg/l chromium in the form Na_2CrO_4 for the shore crab (*Carcinas maenus*), and 1 mg/l for the polychaete *Nereis virens*. Pringle et al. (1968)[250] showed that chromium concentrations of 0.1 and 0.2 mg/l, in the form of $K_2Cr_2O_7$, produced the same mortality with molluscs as the controls. Doudoroff and Katz (1953)[150] investigated the effect of $K_2Cr_2O_7$ on mummichogs (*Fundulus heteroclitus*) and found that they tolerated a concentration of 200 mg/l in sea water for over a week.

Holland et al. (1960)[192] reported that 31.8 mg/l of chromium as potassium chromate in sea water gave 100 per cent mortality to coho salmon (*Oncorhynchus kisutch*). Gooding (1954)[173] found that 17.8 mg/l of hexavalent chromium was toxic to the same species in sea water.

Clendenning and North (1960)[141] showed that hexavalent chromium at 5.0 mg/l chromium reduced photosynthesis in the giant kelp (*Macrocystis pyrifera*) by 50 per cent during 4 days exposure.

Recommendation

Because of the sensitivity of lower forms of aquatic life to chromium and its accumulation at all trophic levels, it is recommended that an application factor of 0.01 be applied to marine 96-hour LC50 data for the appropriate organisms most sensitive to chromium. On the basis of data available at this time, it is suggested that concentrations of chromium equal to or exceeding 0.1 mg/l constitute a hazard to the marine environment, and levels less than 0.05 mg/l present minimal risk of deleterious effects. In oyster areas, concentrations should be maintained at less than 0.01 mg/l.

Copper

Copper has been used as a pesticide for eliminating algae in water, and its salts have bactericidal properties. Copper is toxic to invertebrates and is used extensively in marine antifouling paints which release it to the water. It is also toxic to juvenile stages of salmon and other sensitive species

(Sprague 1964,[267], 1965,[268] Sigler et al. 1966,[263] Cope 1966[142]).

Copper was the fifth metal in U.S. consumption during 1968, following iron, manganese, zinc, and barium (U.S. Department of the Interior Bureau of Mines 1969).[289] Copper is used for such products as high transmission wires, containers, utensils, and currency because of its noncorroding properties.

Copper is widely distributed in nature and is present in sea water in concentrations ranging from 1 to 25 μg/l. In small amounts, copper is nonlethal to aquatic organisms; in fact, it is essential to some of the respiratory pigments in animals (Wilber 1969).[290] Copper chelated by lignin or citrate has been reported to be as effective as copper ion in controlling algae, but apparently it is not as toxic to fish (Ingols 1955).[196] Copper affected the polychaete *Nereis virens* at levels of approximately 0.1 mg/l (Raymont and Shields 1964)[253] and the shore crab (*Carcinus maenas*) at 1 to 2 mg/l (Wilber 1969).[290] Copper at concentrations of 0.06 mg/l inhibited photosynthesis of the giant kelp (*Macrocystis pyrifera*) by 30 per cent in 2 days and 70 per cent in 4 days (Clendenning and North 1960).[131]

Copper is toxic to some oysters at concentrations above 0.1 mg/l (Galtsoff 1932)[168] and lethal to oysters at 3 mg/l (Wilber 1969).[290] The American oyster (*Crassostrea virginica*) is apparently more sensitive to copper than the Japanese species (*Crassostrea gigas*) (Reish 1964).[254] The 96-hour LC50 for Japanese oysters exposed to copper has been reported as 1.9 mg/l (Fujiya 1960).[165] However, oysters exposed to concentrations as low as 0.13 mg/l turn green in about 21 days (Galtsoff 1932).[168] Although such concentrations of copper are neither lethal to the oysters nor, apparently, harmful to man, green oysters are unmarketable because of appearance. Therefore, in the vicinity of oyster grounds, the recommendation for maximum permissible concentrations of copper in the water is based on marketability, and it is recommended that copper not be introduced into areas where shellfish may be contaminated or where seaweed is harvested.

Copper acts synergistically when present with zinc (Wilber 1969),[290] zinc and cadmium (LaRoche 1972),[211] mercury (Corner and Sparrow 1956),[133] and with pentachlorophenate (Cervenka 1959).[127] Studies on sublethal effects of copper show that Atlantic salmon (*Salmo salar*) will avoid concentrations of 0.0024 mg/l in laboratory experiments (Sprague et al. 1965,[270] Saunders and Sprague 1967,[257] Sprague 1971[269]).

Copper is accumulated by marine organisms, with concentration factors of 30,000 in phytoplankton, 5,000 in the soft tissues of molluscs, and 1000 in fish muscle (Lowman et al. 1971).[221]

Bryan and Hummerstone (1971)[124] reported that the polychaete *Nereis diversicolor* shows a high takeup of copper from copper-rich sediments and develops a tolerance. Mobile predators feeding on this species could receive doses toxic to themselves or accumulate concentrations that would be toxic to higher trophic levels.

Recommendation

It is recommended that an application factor of 0.01 be applied to marine 96-hour LC50 data for the appropriate organisms most sensitive to copper. On the basis of data available at this time, it is suggested that concentrations of copper equal to or exceeding 0.05 mg/l constitute a hazard in the marine environment, and levels less than 0.01 mg/l present minimal risk of deleterious effects.

Cyanides

Most of the available information on toxicity of cyanides is for freshwater organisms, and is discussed in the Freshwater Aquatic Life and Wildlife section, p. 189.

Recommendation

As a guideline in the absence of data for marine organisms the panel recommends that an application factor of 0.1 be applied to marine 96-hour LC50 data for the appropriate organisms most sensitive to cyanide. On the basis of data available at this time it is suggested that concentrations of cyanide equal to or exceeding 0.01 mg/l constitute a hazard in the marine environment, and levels less than 0.005 mg/l present minimal risk of deleterious effects.

Fluorides

Fluorides have been brought to public attention in recent years because of their effects at low concentrations in human dental development and in prevention of decay. However, it must be remembered that fluorides at higher concentrations are poisons afflicting human and other mammalian skeletal structures with fluorosis (see Section II, p. 66).

Fluorine is the most reactive non-metal and does not occur free in nature. It is found in sedimentary rocks as fluorspar, calcium fluoride, and in igneous rocks as cryolite, sodium aluminum fluoride. Seldom found in high concentrations in natural surface waters because of their origin only in certain rocks in certain regions, fluorides may be found in detrimental concentrations in ground waters.

Fluorides are emitted to the atmosphere and into effluents from electrolytic reduction plants producing phosphorus and aluminum. They are also used for disinfection, as insecticides, as a flux for steel manufacture, for manufacture of glass and enamels, for preserving wood, and for assorted chemical purposes.

A review of fluoride in the environment (Marier and Rose 1971)[225] indicates that the concentration of unbound ionic fluoride (F^-) in sea water ranges between 0.4 and 0.7 mg/l. Approximately 50 per cent of the total seawater fluoride (0.77 to 1.40 mg/l) is bound as the double ion MgF^+.

Concentrations as low as 1.5 mg/l of fluoride have affected hatching of fish eggs (Ellis et al. 1946),[157] and 2.3 mg/l, introduced as sodium fluoride, was lethal to rainbow trout (*Salmo gairdneri*) at 18 C (Angelovic et al. 1961).[113] Virtually no information exists on long-term chronic effects of low concentrations of fluorides in sea water.

Recommendation

In the absence of data on the sublethal effects of fluorides in the marine environment, it is recommended that an application factor of 0.1 be applied to marine 96-hour LC50 data for the appropriate organisms most sensitive to fluoride. On the basis of data available at this time it is suggested that concentrations of fluoride equal to or exceeding 1.5 mg/l constitute a hazard in the marine environment, and levels less than 0.5 mg/l present minimal risk of deleterious effects.

Iron

Because of the widespread use of iron by man for his many industrial activities, iron is a common contaminant in the aquatic environment. Iron may enter water naturally from iron ore deposits; but iron is more often introduced from acid mine drainage, mineral processing, steel pickling, and corrosion. Iron usually occurs in the ferrous form, when it is released from processing plants or in mine drainage, but becomes rapidly oxidized to the ferric form in natural surface waters. The ferric salts form gelatinous hydroxides, agglomerate and flocculate, settling out on the bottom or becoming adsorbed on various surfaces. Depending on the pH and Eh, groundwater may contain a considerable amount of iron in solution, but well aerated waters seldom contain high, dissolved iron. In the marine environment, iron is frequently present in organic complexes and in adsorbed form on particulate matter.

Most of the investigations on biological effects of iron have been done in fresh water (Knight 1901,[204] Bandt 1948,[117] Minkina 1946,[229] Southgate 1948,[265] Lewis 1960,[215] ORSANCO 1960[245]). Deposition of iron hydroxides on spawning grounds may smother fish eggs, and the hydroxides may irritate the gills and block the respiratory channels of fishes (Southgate 1948,[265] Lewis 1960[215]). Direct toxicity of iron depends on its valence state and whether it is in solution or suspension.

Warnick and Bell (1969)[297] examined the effects of iron on mayflies, stoneflies, and caddisflies and obtained a 96-hour LC50 of 0.32 mg/l for the three insects. Dowden and Bennett (1965)[153] examined the effect of ferric chloride to *Daphnia magna* in static acute bioassays. They noted LC50's of 36, 21, and 15 mg/l for 1, 2, and 4 days, respectively.

Ferric hydroxide flocs removed the diatoms in the process of flocculation and settling, coating the bottom; and the iron precipitate coated the gills of white perch (*Morone americana*), minnows, and silversides in upper Chesapeake Bay (Olson et al. 1941).[242]

Tests on three types of fish gave a lethality threshold for iron at 0.2 mg/l (Minkina 1946)[229] and on carp at 0.9 mg/l if the pH was 5.5 or lower. Ebeling (1928)[155] found that 10 mg/l of iron caused serious injury or death to rainbow trout (*Salmo gairdneri*) in 5 minutes. La Roze (1955)[212] reported that dogfish were killed in 3 hours at 5 mg/l iron, whereas other research (National Council for Stream Improvement 1953)[236] indicated no deaths during one week at 1 to 2 mg/l.

Because of the slightly alkaline condition of sea water, much of the iron introduced to the sea precipitates. This adds a further problem of iron hydroxide flocs contaminating bottom sediments where rooted aquatics and invertebrates could be affected.

Special consideration should be given to avoiding discharge of iron-containing effluents into waters where commercially important bottom species or important food organisms dwell (e.g., oysters, clams, scallops, lobsters, crabs, shrimp, halibut, flounder, and demersal fish eggs and larvae).

Recommendation

On the basis of data available at this time, it is suggested that concentrations of iron equal to or exceeding 0.3 mg/l constitute a hazard to the marine environment, and levels less than 0.05 mg/l present minimal risk of deleterious effects.

Lead

The present rate of input of lead into the oceans is approximately ten times the rate of introduction by natural weathering, and concentrations of lead in surface sea water are greater than in deeper waters (Chow and Patterson 1966).[139] The isotope composition of the lead in surface waters and in recent precipitation is more similar to that of mined ore than to that in marine sediments (Chow 1968).[138] There are almost no data, however, that would suggest that the higher concentrations of lead in surface sea water derived from lead transported through the atmosphere have resulted in higher lead concentrations in marine wildlife. Lead concentrations in Greenland snow have been shown to be 16 times higher in 1964 than in 1904 (Murozumi et al. 1969).[235] In 1968 an estimated 1.8×10^5 tons of lead were introduced to the atmosphere as a result of the combustion of leaded gasoline (Council on Environmental Quality 1971).[144] This represents 14 per cent of the total lead consumption of the United States for that year. Lead poisoning of zoo animals in New York City was attributed to their breathing lead-contaminated air (Bazell 1971).[119]

Blood serum aldolase activity in higher animals exposed to small amounts of lead increased, although there were no overt signs or symptoms of poisoning (Yaverbaum 1963,[308] Wilber 1969[299]). Chronic lead poisoning in man is symptomatically similar to multiple sclerosis (Falkowska et al. 1964).[159] Muscular dystrophy has been reported as occurring in fishes and amphibians (Stolk 1962,[272] Wilber 1969[299]) and in view of these findings could, in fact, be unnatural.

Data are needed on the sublethal, long-term effects of lead on aquatic organisms, particularly those in sea water. Evidence of deleterious effect to freshwater fish has been reported for concentrations of lead as low as 0.1 mg/l (Jones 1938).[197]

Wilder (1952)[300] reported lobster dying in 6 to 20 days when held in lead-lined tanks. Pringle (*unpublished data*)[315] observed a 12-week LC50 of 0.5 mg/l lead and an 18-week LC50 of 0.3 mg/l lead with the oyster (*Crassostrea virginica*). There was noticeable change in gonadal and mantle tissue following 12 weeks exposure at concentrations of 0.1 to 0.2 mg/l of lead. Calabrese et al. (*unpublished data*)[310] found a 48-hour LC25 of 1.73 mg/l and an LC50 of 2.45 mg/l for oyster eggs of the same species.

North and Clendenning (1958)[241] reported that lead nitrate at 4.1 mg/l of lead showed no deleterious effect on the photosynthesis rate in kelp (*Macrocystis pyrifera*) exposed for four days. They concluded that lead is less toxic to kelp than mercury, copper, hexavalent chromium, zinc, and nickel.

Recommendation

In the absence of more definitive information on the long-term chronic effect of lead on marine organisms, it is recommended that concentrations of lead in sea water should not exceed 0.02 of the 96-hr LC50 for the most sensitive species, and that the 24-hour average concentration should not exceed 0.01 of the 96-hour LC50. On the basis of data available at this time it is suggested that concentrations of lead equal to or exceeding 0.05 mg/l constitute a hazard in the marine environment, and levels less than 0.01 mg/l present minimal risk of deleterious effects. Special effort should be made to reduce lead levels even further in oyster-growing areas.

Lead recommendations for the protection of wildlife are included in the discussion of Marine Wildlife p. 227.

Manganese

Manganese is one of the most commonly used metals in industry. It occurs widely in ores on land and in nodules in the deep sea. U.S. consumption in 1958 exceeded 2.2 million tons, a 45 per cent increase in 20 years (U.S. Department of Interior, Bureau of Mines 1969).[289] The metal is alloyed with iron to produce steel and in smaller quantities with copper for manganese bronze. Its salts are used in inks and dyes, in glass and ceramics, in matches and fireworks, for dry-cell batteries, and in the manufacture of paints and varnishes.

Manganese is often found with iron in ground waters, and it can be leached from soil and occur in drainage in high concentrations. The carbonates, oxides, and hydroxides are slightly soluble, so that manganous and manganic ions are rarely present in surface water in excess of 1 mg/l. Manganese is present in sea water at about 2 μg/l in the Mn^{+2} form, and is concentrated through biochemical processes to form manganese nodules, found mainly in the deep sea.

Manganese may have different effects on the lower trophic levels in fresh water and sea water. Concentrations of manganese above 0.005 mg/l had a toxic effect on certain algae in reservoirs (Guseva 1937,[174] 1939[175]), while 0.0005 mg/l in sea water stimulated growth and multiplication of certain phytoplankton (Harvey 1947).[178] Anderson (1944)[110] reported the threshold of immobilization of *Daphnia magna* as 0.63 mg/l of $KMnO_4$ and the threshold concentration for immobilization of *Daphnia magna* in Lake Erie water as 50 mg/l of $MnCl_2$ (Anderson 1948).[112] Bringmann and Kuhn (1959a)[131] reported the threshold effect for the same species as 50 mg/l of $MnCl_2$ as manganese in River Havel water at 23 C.

For the flatworm *Polycelis nigra*, the threshold concentration of manganese was reported as 700 mg/l as manganese chloride and 660 mg/l as manganese nitrate (Jones 1940).[199] Tests on organisms on which fish feed, i.e., crustacea, worms, and insect larvae, showed no apparent harm at 15 mg/l of manganese during a 7-day exposure (Schweiger 1957).[261] River crayfish were found to tolerate 1 mg/l (Meinck et al. 1956).[227]

The toxicity of manganese to fish depends on a number of factors which may vary from one situation to another. There is an apparent antagonistic action of manganese toward nickel toxicity for fish (Blabaum and Nichols 1956).[125] This may be true also for cobalt and manganese in combination, as noted for terrestrial plant life (Ahmed and Twyman 1953).[108]

Stickleback survived 50 mg/l manganese as manganese sulphate for 3 days, whereas eels withstood 2700 mg/l for 50 hours (Doudoroff and Katz 1953).[150] The lethal concentration of manganese for stickleback was given as 40 mg/l by Jones (1939),[198] and he noted that the toxic action was slow. The minimum lethal concentration of manganese nitrate for sticklebacks in tap water has been reported to be 40 mg/l as manganese (Anderson 1948,[112] Murdock 1953).[234]

The average survival times of stickelback in manganous nitrate solution were one week at 50 mg/l, four days at 100 mg/l, two days at 150 mg/l, and one day at 300 mg/l, all measured as manganese (Murdock 1953).[234] Young eels tolerated 1500 mg/l manganous sulphate for more than 25 hours (Doudoroff and Katz 1953).[150] Oshima (1931)[246] and Iwao (1936)[189] reported the lethal thresholds of manganous chloride and manganous sulphate for fish in Japan to be about 2400 and 1240 mg/l of manganese, respectively. They found that permanganates (Mn^{+7}) killed fish at 2.2 to 4.1 mg/l manganese in 8 to 18 hours, but this high oxidation form is quite unstable in water. Tench, carp, and trout tolerated 15 mg/l of manganese during 7 days exposure (Schweiger 1957).[261]

Manganous chloride was found to be lethal to minnows (*Fundulus*) in fresh water in six days at 12 mg/l $MnCl_2$; for the small freshwater fish Orizias, the 24 hour lethal concentration was about 7850 mg/l (Doudoroff and Katz 1953);[150] and for other fish 5500 mg/l (Oshima 1931,[246] Iwao, 1936[189]). The highest concentration tolerated by eels for 50 hours was 6300 mg/l (Doudoroff and Katz 1953).[150] Meinck et al. (1956)[227] noted the first toxic effects for fish of $MnCl_2$ at 330 mg/l, with the lethal concentration at 800 mg/l.

Only a few studies of sublethal effects of manganese on fish have been reported. Ludemann (1953)[222] noted some of the symptoms of toxicity of manganese to fish, crabs, and fish food organisms. Abou-Donia and Menzel (1967)[103] noted an effect of 1.25×10^{-4} M manganese (6.9 mg/l) on the enzyme acetylcholinesterase in shiner perch.

In studies on the uptake of radionuclides on the Pacific testing grounds of Bikini and Eniwetok, it was found that the neutron-induced isotope of manganese ^{54}Mn was concentrated by as much as 4000 in phytoplankton and 12,000 in the muscle or soft tissue of mollusks (Lowman 1960,[220] Lowman et al. 1971[221]). Goldberg et al. (1971)[172] list the concentration factor of manganese in marine plants and animals as approximately 3000.

Recommendation

In view of the evidence for concentration of manganese by marine organisms, an application factor of 0.02 of the 96 hr LC50 for the most sensitive species to be protected is recommended.

Until more complete information on acute and sublethal effects of manganese on marine organisms is available, it is suggested that concentrations of 0.1 mg/l or more of total manganese in the marine environment may constitute a hazard, and concentrations of less than 0.02 mg/l present minimal risk.

Mercury

Mercury naturally leaches from cinnabar (HgS) deposits. Man-made sources of mercury have been in plastics manufacture, where mercury oxide is used as a catalyst, chlor-alkali plants where mercury cells are used, mercurial slimacides used in the pulp and paper industry and in other forest product anti-fungal applications, seed dressings used in combatting smuts and other fungal diseases afflicting seeds, and in anti-fouling paints. An estimated 5000 tons of mercury per year are transferred from the continents to the oceans as a result of continental weathering (Klein and Goldberg 1970).[203] Global production of mercury is currently about twice as high, in the order of 9000 metric tons per year (Hammond 1971).[176] The burning of petroleum releases in the order of 1600 tons of mercury into the atmosphere per year (Bertine and Goldberg 1971).[122] A conservative estimate of the amount of mercury released per year into the global environment from the burning of coal is in the order of 3000 tons (Joensuu 1971).[195] The total amount of mercury estimated to be in the oceans is in the order of 10^8 metric tons, approximately three orders of magnitude higher than the total amount of mercury consumed in the United States since 1900. Mercury in marine organisms is, therefore, most probably of natural origin except in localized areas.

One hundred and eleven persons were reported poisoned, 41 died, and others suffered serious neurological damage as a result of eating fish and shellfish which had been contaminated with mercury discharged into Minamata Bay by a plastics manufacturing plant between 1950 and 1960 (Irukayama 1967).[187] In 1965, another poisoning incident was reported in Niigata, Japan, where 5 people died and 26 suffered irreversible neurological damage (Ui 1967).[292] In Minamata it was also found that cats eating the contaminated fish and shellfish took suicidal plunges into the sea, an uncommon occurrence with these mammals (Ui and Kitamura 1970).[293]

Metallic mercury can be converted by bacteria into methyl mercury (Jensen and Jernelöv 1969,[192] Jernelöv 1969,[194] Löfroth 1969[218]). Organometallic mercury is much more toxic than the metallic mercury and enters the food cycle through uptake by aquatic plants, lower forms of animal life, and fish (Jernelöv 1969).[194] The concentration factor of mercury in fish was reported as 3,000 and higher (Hannerz 1968,[177] Johnels and Westermark 1969[196]). A voluntary form of control was imposed in Sweden where anglers were requested not to eat more than one fish per week from a given lake to minimize human intake.

High mercury concentrations in birds and fish were reported on the Canadian prairies in 1969 (Fimreite 1970,[160] Wobeser et al. 1970,[301] Bligh 1971[126]). The source of the mercury in the birds was apparently mercurial seed dressing consumed with grain by the birds; whereas in fish, mercury came largely from emissions of a chlor-alkali plant using a mercury cell.

The Food and Drug Directorate of Canada set a level of 0.5 parts per million as the maximum permissible concentration in fish products. The 0.5 parts per million level was set as an interim guideline, not a regulation based on any known safe level for mercury (Canada Food and Drug Directorate, *personal communication*).[311] A similar guideline was adopted in the U.S. (Kolbye 1970).[206] These limits were based on the lethal concentrations found in Minamata Bay, Japan, and on the levels set by the World Health Organization (WHO) in cooperation with the Food and Agricultural Organization (FAO). The level set by WHO/FAO was 0.05 ppm, based on total food (WHO 1967).[305] The concentrations which were found lethal to the Japanese consuming fish and shellfish contaminated by mercury were 10 to 50 mg/kg total mercury (Birke et al. 1968).[124] The Swedish limit was 1.0 ppm of mercury in fish (Berglund and Wretling 1967)[121] based on dry weight, which is equivalent to 0.2 ppm wet weight (Wallace et al. 1971).[295]

Although the emphasis has been on the effects of mercury on man, aquatic organisms can be affected by various mercury compounds. Mercury markedly alters the epithelium of skin and gills in fishes (Schweiger 1957).[261] Mercuric chloride in water containing developing eggs of *Paracentrotus lividis* brought about a severe disturbance of development at 10 μg/l (Soyer 1963).[266] A concentration of 5 μg/l retarded development markedly. These studies suggested that the threshold for harmful effects of mercuric chloride on developing eggs of *Paracentrotus* was around 2 to 3 μg/l (Soyer 1963).[266] Studies conducted on developing salmon eggs (*Oncorhynchus nerka* and *O. gorbuscha*) at the International Pacific Salmon Fisheries Commission Laboratory in Cultus Lake, B.C., showed that concentrations of mercury at levels exceeding 3 μg/l mercury derived from mercuric sulfate led to severe deformities (Servizi, *unpublished data*).[316] Studies are needed to examine the effects of those concentrations which are accumulated by fish over a longer period of time.

Ukeles (1962)[253] reported that 60 μg/l of ethyl mercury phosphate was lethal to all species of marine phytoplankton tested, and that as little as 0.1 to 0.6 μg/l of alkyl mercury introduced into sea water will inhibit photosynthesis and growth. Clendenning and North (1960)[141] reported that mercury added as mercuric chloride caused 50 per cent inactivation of photosynthesis of giant kelp (*Macrocystis pyrifera*) at 50 μg/l during 4 days exposure, a 15 per cent decrease in photosynthesis at 100 μg/l in 1 day, and complete inactivation in 4 days.

Woelke (1961)[302] reported that 27 μg/l of mercury as mercuric chloride was lethal to bivalve larvae. The learning behavior of goldfish (*Carassius auratus*) was affected after two days by 3 μg/l mercuric chloride (Weir and Hine 1970).[298] Trace amounts of copper increase the toxicity of mercury (Corner and Sparrow 1956).[143]

Mercury concentrations in tissues of the Ashy Petrel (*Oceanodroma homochroa*) from the coastal waters of California, the site of most of the mercury mines in the United States, are in the same order of magnitude as mercury concentrations in tissues of the Snow Petrel (*Pelagodroma nivea*), which inhabits the Antarctic pack ice. Mercury concentrations in nine eggs of the Common Tern (*Sterna hirundo*) from Long Island Sound were only slightly higher than in nine eggs of the Antarctic Tern (*Sterna vittata*) from the Antarctic (Anderlini et al. *in press*).[109]

Environmental residues of mercury in Sweden, as measured by concentrations of mercury in feathers of several species of birds, rose dramatically in the years following 1940 and were attributed to alkyl-mercury compounds used as seed dressings (Berg et al. 1966).[120] This use of mercury caused the death of numbers of seed-eating birds (Borg et al. 1969),[130] but it does not necessarily contaminate aquatic ecosystems (Johnels and Westermark 1969).[196] Feathers of two species of fish-eating birds, the Osprey (*Pandion haliaetus*) and the Great-crested Grebe (*Pocideps cristatus*), have shown a gradual increase in mercury concentration since approximately 1900, paralleling the increase in industrial use of mercury in Sweden (Johnels and Westermark 1969).[196] Experimental work in Sweden has shown that when pheasants were fed wheat treated with methyl-mercury dicyandiamide, decreased hatchability of eggs was associated with mercury concentrations in the eggs from 1.3 to 2.0 mg/kg of the wet weight contents (Borg et al. 1969).[130] It has been suggested that environmental mercury may impair the reproductive capacity of bird species at the tops of food chains, such as falcons (Fimreite et al. 1970),[161] and in Finland mercury may have contributed to the decline of the Whitetailed Sea Eagle (*Haliaetus albicilla*) in regions where the species feeds upon marine fish and marine birds (Henriksson et al. 1966).[180] Conclusive evidence that mercury has impaired the reproductive capacity of any species of wildlife, however, has not yet been obtained and further research is necessary. Fish-eating birds and mammals are the species most likely to be affected because of their position at the top of the food chain.

The high natural levels of mercury in the marine environment and the significant additions due to natural weathering, as well as the documented hazard to marine aquatic life and to humans through marine foods, make it desirable to eliminate inputs of mercury to the marine environment beyond those occurring through continental weathering.

Recommendation

On the basis of data available at this time, it is suggested that concentrations of mercury equal to or exceeding 0.10 μg/l constitute a hazard in the marine environment.

In the absence of sufficient data on the effects of mercury in water upon wildlife, the recommendations established to protect aquatic life and public water supplies should also apply to protect wildlife.

Molybdenum

Molybdenum has been found to be a needed microconstituent in fresh waters for normal growth of phytoplankton (Arnon and Wessel 1953).[115] In mammals, exposure to molybdenum may interfere with vital chemical reactions (Dick and Ball 1945).[146]

Molybdenum metal is quite stable and is used in ferro-molybdenum for the manufacture of special tool steels. It is available in a number of oxide forms as well as the disulphide. Molybdic acid is used in a number of chemical applications and in make-up of glazes for ceramics.

Molybdenum has not been considered as a serious pollutant, but it is a biologically active metal. It may be an important element insofar as protection of the ecosystem is concerned because of its role in algal physiology. Certain species of algae can concentrate molybdenum by a factor up to 15 (Lackey 1959).[210] Bioassay tests in fresh water on the fathead minnow gave a 96-hour LC50 for molybdic anhydride (MoO_3) of 70 mg/l in soft water and 370 mg/l in hard water. Although molybdenum is essential for the growth of the alga *Scenedesmus*, the threshold concentration for a deleterious effect is 54 mg/l. Molybdenum concentration factors for marine species have been reported as: 8 in benthic algae; 26 in zooplankton; 60 in soft parts of molluscs; 10 in crustacean muscles; and 10 in fish muscle (Lowman et al. 1971).[221]

Recommendation

The panel recommends that the concentration of molybdenum in sea water not exceed 0.05 of the 96-hour LC50 at any time for the most sensitive species in sea water, and that the 24-hour average not exceed 0.02 of the 96-hour LC50.

Nickel

Nickel does not occur naturally in elemental form. It is present as a constituent in many ores, minerals and soils, particularly in serpentine-rock-derived soils.

Nickel is comparatively inert and is used in corrosion-resistant materials, long-lived batteries, electrical contacts, spark plugs, and electrodes. Nickel is used as a catalyst in hydrogenation of oils and other organic substances. Its salts are used for dyes in ceramic, fabric, and ink manufacturing. Nickel may enter waters from mine wastes, electroplating plants, and from atmospheric emissions.

Nickel ions are toxic, particularly to plant life, and may exhibit synergism when present with other metallic ions. Nickel salts in combination with a cyanide salt form moderately toxic cyanide complexes which, as nickel sulfate combined with sodium cyanide, gave a 48-hour LC50 of 2.5 mg/l and a 96-hour LC50 of 0.95 mg/l as CN^-, using fathead minnows (*Pimephales promelas*) at 20 C (Doudoroff 1956).[148] Alkaline conditions reduced toxicity of a nickel cyanide complex considerably, with concentrations below 100 mg/l showing no apparent toxic effect on fish.

Nickel salts can substantially inhibit the biochemical oxidation of sewage (Malaney et al. 1959).[223] In fresh waters, nickel has been reported to be less toxic to fish and river crabs than zinc, copper, and iron (Podubsky and Stedronsky 1948).[248] However, other investigators found nickel to be more toxic to fish than iron and manganese (Doudoroff and Katz 1953).[150]

Ellis (1937)[156] reported that nickelous chloride from electroplating wastes did not kill goldfish (*Carassius auratus*) at 10 mg/l during a 200-hour exposure in very soft water. Wood (1964)[304] reported that 12 mg/l of nickel ion kill fish in 1 day and 0.8 mg/l kill fish in 10 days. Doudoroff and Katz (1953)[150] reported survival of stickleback (*Gasterosteus aculeatus*) for 1 week in 1 mg/l of nickel as $Ni(NO_3)_2$.

The lethal limit of nickel to sticklebacks has been reported as 0.8 mg/l (Murdock 1953)[234] and 1.0 mg/l (Jones 1939).[198] The median lethal concentration of nickel chloride ($NiCl_2 , 6H_2O$) was reported as 4.8 mg/l for guppies (*Becilia reticulata*) (Shaw and Lowrance 1956).[262] Goldfish (*Carassius auratus*) were killed by nickel chloride at 4.5 mg/l as nickel in 200 hours (Rudolfs et al. 1953).[256] Tarzwell and Henderson (1960)[278] reported 96-hour LC50's for fathead minnows (*Pimephales promelas*) as 4.0 mg/l in soft water and 24 mg/l in hard water, expressed as $NiCl_2 , 6H_2O$. Anderson (1948)[112] reported a threshold concentration of nickel chloride for immobilization of *Daphnia* in Lake Erie water at 25 C to be less than 0.7 mg/l in 64 hours of exposure. Bringmann and Kuhn (1959a,[131] 1959b[132]) reported nickel chloride threshold concentrations as nickel of 1.5 mg/l for *Scenedesmus*, 0.1 mg/l for *Escherichia coli*, and 0.05 mg/l for *Microregma*.

Nickel is present in sea water at 5 to 7 µg/l, in marine plants at up to 3 mg/l, and in marine animals at about 0.4 mg/l.

Marine toxicity data for nickel are limited. The top minnow *Fundulus* was found to survive in concentrations of 100 mg/l Nickel from the chloride in salt water, although the same species was killed by 8.1 mg/l of the salt (3.7 mg/l Ni) in tap water (Thomas cited by Doudoroff and Katz 1953).[150] Long-term studies on oysters (Haydu *unpublished data*)[314] showed substantial mortality at a nickel concentration of 0.12 mg/l. Calabrese et al. (*unpublished data*)[310] found 1.54 mg/l of nickel to be the LC50 for eggs of the oyster (*Crassostrea virginica*).

Recommendation

It is recommended that an application factor of 0.02 be applied to 96-hour LC50 data on the most sensitive marine species to be protected. Although limited data are available on the marine environment, it is suggested that concentrations of nickel in excess of 0.1 mg/l would pose a hazard to marine organisms, and 0.002 mg/l should pose minimal risk.

Phosphorus

Phosphorus as phosphate is one of the major nutrients required for algal nutrition. In this form it is not normally toxic to aquatic organisms or to man. Phosphate in large quantities in natural waters, particularly in fresh waters, can lead to nuisance algal growths and to eutrophication. This is particularly true if there is a sufficient amount of nitrate or other nitrogen compounds to supplement the phosphate. Thus, there is a need for control of phosphate input into marine waters. See Sewage and Nutrients, p. 275, for a discussion of the effects of phosphate as a nutrient.

Phosphorus in the elemental form is particularly toxic and subject to bioaccumulation in much the same way as mercury (Ackman et al. 1970,[104] Fletcher 1971[162]). Isom (1960)[188] reported an LC50 of 0.105 mg/l at 48 hours and 0.025 mg/l at 163 hours for bluegill sunfish (*Lepomis macrochirus*) exposed to yellow phosphorus in distilled water at 26 C and pH 7.

Phosphorus poisoning of fish occurred on the coast of Newfoundland in 1969 and demonstrated what can happen when the form of an element entering the sea is unknown or at least not properly recognized (Idler 1969,[185] Jangaard 1970,[191] Mann and Sprague 1970[224]). The elemental phosphorus was released in colloidal form and remained in suspension (Addison and Ackman 1970).[105] After the release of phosphorus was initiated, red herrings began to appear. The red discoloration was caused by haemolysis, typical of phosphorus poisoning in herring (*Clupea harengus*), and elemental phosphorus was found in herring, among other fishes, collected 15 miles away (Idler 1969,[185] Jangaard 1970[191]).

Fish will concentrate phosphorus from water containing as little as one µg/l (Idler 1969).[185] In one set of experiments, a cod swimming in water containing one µg/l elemental phosphorus for 18 hours was sacrificed and the tissues analyzed. The white muscle contained about 50 µg/kg, the brown, fat tissue about 150 µg/kg, and the liver 25,000 µg/l (Idler 1969,[185] Jangaard 1970[191]). The experimental findings showed that phosphorus is quite stable in the fish tissues. Fish with concentrated phosphorus in their tissues could swim for considerable distances before succumbing. In addition to the red surface discoloration in herring, other diagnostic features of phosphorus poisoning included green discoloration of the liver and a breakdown of the epithelial lining of the lamellae of the gill (Idler 1969).[185]

A school of herring came into the harbor one and one-half months after the phosphorus plant had been closed down. These herring spawned on the wharf and rocks near the effluent pipe, and many of them turned red and died. A few days later, "red" herring were caught at the mouth of the harbor on their way out. The herring picked up phosphorus from the bottom sediments which contained high concentrations near the effluent pipeline (Ackman et al. 1970).[104] Subsequently, this area was dredged by suction pipeline, and the mud was pumped to settling and treatment ponds. No further instances of red herring were reported after the dredging operation, and the water was comparatively free of elemental phosphorus (Addison et al. 1971).[106]

Reports of red cod caught in the Placentia Bay area were investigated, and it was found that no phosphorus was present in the cod tissues. Surveys of various fishing areas in Newfoundland established that red cod are no more prevalent in Placentia Bay than in other areas. In laboratory studies, cod exposed to elemental phosphorus have not shown the red discoloration observed in herring and salmonids. However, cod do concentrate phosphorus in the muscle tissue as well as in the liver and can eventually succumb to phosphorus poisoning (Dyer et al. 1970).[154]

It was demonstrated by field investigations and laboratory experiments (Ackman et al. 1970,[104] Fletcher et al. 1970,[163] Li et al. 1970,[216] Zitko et al. 1970,[309] Fletcher 1971[162]) that elemental phosphorus accounted for the fish mortalities in Placentia Bay. This is not to say that other pollutants, such as fluorides, cyanides, and ammonia, were not present (Idler 1969).[185]

The conclusion was reached by the scientists working on the problem that elemental phosphorus in concentrations so low that they would be barely within the limits of detection are capable of being concentrated by fish. Further work is needed on the effects of very low concentrations of phosphorus on fish over extended periods. Discharge of elemental phosphorus into the sea is not recommended.

Recommendation

It is recommended that an application factor of 0.01 be applied to marine 96-hour LC50 data for the appropriate organisms most sensitive to elemental phosphorus. On the basis of data available at this time it is suggested that concentrations of elemental phosphorus equal to or exceeding 1 µg/l constitute a hazard to the marine environment.

Selenium

Selenium has been regarded as one of the dangerous chemicals reaching the aquatic environment. Selenium occurs naturally in certain pasture areas. Toxicity of selenium is sometimes counteracted by the addition of arsenic which acts as an antagonist. Selenium occurs in nature chiefly in combination with heavy metals. It exists in several forms including amorphous, colloidal, crystalline, and grey. Each physical state has different characteristics, soluble in one form, but insoluble in another. The crystalline and grey forms conduct electricity, and the conductivity is increased by light. This property makes the element suitable for photoelectric cells and other photometry uses. Selenium is also used in the manufacture of ruby glass, in wireless telegraphy and photography, in vulcanizing rubber, in insecticidal preparations, and in flameproofing electric cables. The amorphous form is used as a catalyst in determination of nitrogen and for dehydrogenation of organic compounds.

Ellis (1937)[156] showed that goldfish (*Carassius auratus*) could survive for 98 to 144 hours in soft water of pH ranging from 6.4 to 7.3 at 10 mg/l sodium selenite. Other data (ORSANCO 1950)[243] showed that 2.0 mg/l of selenium administered as sodium selenite was toxic in 8 days, affecting appetite and equilibrium, and lethal in 18 to 46 days.

More work is required to test for effects of selenium compounds under different conditions. Daphnia exhibited a threshold effect at 2.5 mg/l of selenium in a 48-hour exposure at 23 C (Bringmann and Kuhn 1959a).[131] Barnhart (1958)[118] reported that mortalities of fish stocked in a Colorado reservoir were caused by selenium leached from bottom deposits, passed through the food chain, and accumulated to lethal concentrations by the fish in their liver.

Recommendation

In view of the possibility that selenium may be passed through the food chain and accumulated in fish, it is recommended that an application factor of 0.01 be applied to marine 96-hour LC50 data for the appropriate organisms most sensitive to selenium. On the basis of data available at this time, it is suggested that concentrations of selenium equal to or exceeding 0.01 mg/l constitute a hazard in the marine environment, and levels less than 0.005 mg/l present minimal risk of deleterious effects.

Silver

Silver is one of the more commercially important metals; 4,938 tons were consumed in the U.S. during 1968, excluding that used for monetary purposes (U.S. Department of the Interior, Bureau of Mines 1969).[289] It is the best known conductor of heat and electricity. Although not oxidized by air, silver is readily affected by hydrogen sulfide to form the black silver sulfide.

Silver has many uses. In addition to making currency, it is used for photographic purposes, for various chemical purposes, and also in jewelry making and in silverplating of cutlery.

Silver is toxic to aquatic animals. Concentrations of 400 µg/l killed 90 per cent of test barnacles (*Balanus balanoides*) in 48 hours (Clarke 1947).[140] Concentrations of silver nitrate from 10 to 100 µg/l caused abnormal or inhibited development of eggs of *Paracentrotus* and concentrations of 2 µg/l of silver nitrate delayed development and caused deformation of the resulting plutei (Soyer 1963).[266] Adverse effects occurred at concentrations below 0.25 µg/l of silver nitrate, and several days were required to eliminate adverse effects by placing organisms in clean water (Soyer 1963).[266] Silver nitrate effects on development of *Arbacia* have been reported at approximately 0.5 µg/l (Soyer 1963,[266] Wilber 1969[299]). In combination with silver, copper acts additively on the development of *Paracentrotus* eggs (Soyer 1963).[266] On a comparative basis on studies on Echinoderm eggs (Soyer 1963),[266] silver has been found to be about 80 times as toxic as zinc, 20 times as toxic as copper, and 10 times as toxic as mercury.

Calabrese et al. (*unpublished manuscript*)[310] noted an LC50 of 0.006 mg/l silver for eggs of the American oyster (*Crassostrea virginica*). Jones (1948)[200] reported that the lethal concentration limit of silver, applied as silver nitrate, for sticklebacks (*Gasterosteus aculeatus*) at 15 to 18 C was 0.003 mg/l, which was confirmed approximately by Anderson (1948),[112] who found 0.0048 mg/l to be the toxic threshold for sticklebacks. Jackim et al. (1970)[190] reported adverse effects on the liver enzymes of the killifish *Fundulus heteroclitus* at 0.04 mg/l of silver.

The sublethal responses to silver compounds may be great, in view of the effects on developing eggs; and further research should be conducted on effects of sublethal concentrations of silver compounds by themselves and in combination with other chemicals. The disruption of normal embryology or of nutrition could be of much greater importance than direct mortality in the perpetuation of the species.

Concentrations of silver cannot exceed that permitted by the low solubility product of silver chloride. However, silver complexes may be present, and their effects are unknown.

Recommendation

It is recommended that the concentrations of silver in marine waters not exceed 0.05 of the 96-hour LC50 for the appropriate species most sensitive to silver. On the basis of data available at this time, it is suggested that concentrations of silver equal to or exceeding 5 µg/l constitute a hazard to the marine environment, and levels less than 1 µg/l present minimal risk of deleterious effects.

Sulfides

Sulfides in the form of hydrogen sulfide have the odor of rotten eggs and are quite toxic. Hydrogen sulfide is soluble in water to the extent of 4000 mg/l at 20 C and 1 atmosphere. Sulfides are produced as a by-product in tanneries, chemical plants, and petroleum refineries, and are used in pulp mills, chemical precipitation, and in chemical production. Hydrogen sulfide is produced in natural decomposition processes and in anaerobic digestion of sewage and industrial wastes. Sulfate in sea water is reduced to sulfide in the absence of oxygen. In the presence of certain sulfur-utilizing bacteria, sulfides can be oxidized to colloidal sulfur. At the normal pH and oxidation-reduction potential of aerated sea water, sulfides quickly oxidize to sulfates.

Hydrogen sulfide dissociates into its constituent ions in two equilibrium stages, which are dependent on pH (McKee and Wolf 1963).[226]

The toxicity of sulfides to fish increases as the pH is lowered because of the HS^- or H_2S molecule (Southgate 1948).[265] Inorganic sulfides are fatal to sensitive species such as trout at concentrations of 0.05 to 1.0 mg/l, even in neutral and somewhat alkaline solutions (Doudoroff 1957).[149] Hydrogen sulfide generated from bottom deposits was reported to be lethal to oysters (de Oliveira 1924).[145]

Bioassays with species of Pacific salmon (*Oncorhynchus* *tshawytscha, O. kisutch*) and sea-run trout (*Salmo clarkii clarkii*) showed toxicity of hydrogen sulfide at 1.0 mg/l and survival without injury at 0.3 mg/l (Van Horn et al. 1949,[291] Dimick 1952,[147] Haydu et al. 1952,[179] Murdock 1953,[234] Van Horn 1959[290]). Holland et al. (1960)[182] reported that 1 mg/l of sulfide caused loss of equilibrium in 2 hours, first kills in 3 hours, and 100 per cent mortality in 72 hours with Pacific salmon.

Hydrogen sulfide in bottom sediments can affect the maintenance of benthic invertebrate populations (Thiede et al. 1969).[280] The eggs and juvenile stages of most aquatic organisms appear to be more sensitive to sulfides than do the adults. Adelman and Smith (1970)[107] noted that hydrogen sulfide concentrations of 0.063 and 0.020 mg/l killed northern pike (*Esox lucius*) eggs and fry, respectively; and at 0.018 and 0.006 mg/l, respectively, reduced survival, increased anatomical malformations, or decreased length were reported.

Recommendation

It is recommended that an application factor of 0.1 be applied to marine 96-hour LC50 for the appropriate organisms most sensitive to sulfide. On the basis of data available at this time, it is suggested that concentrations of sulfide equal to or exceeding 0.01 mg/l constitute a hazard in the marine environment, and levels less than 0.005 mg/l present minimal risk of deleterious effects, with the pH maintained within a range of 6.5 to 8.5.

Thallium

Thallium salts are used as poison for rats and other rodents and are cumulative poisons. They are also used for dyes, pigments in fireworks, optical glass, and as a depilatory.

Thallium forms alloys with other metals and readily amalgamates with mercury. It is used in a wide variety of compounds. Nehring (1963)[238] reported that thallium ions were toxic to fishes and aquatic invertebrates. The response of fishes to thallium poisoning is similar to that of man, an elevation in blood pressure. In both the fish and invertebrates, thallium appears to act as a neuro-poison (Wilber 1969).[299]

Adverse effects of thallium nitrate have been reported for rainbow trout (*Salmo gairdneri*) at levels of 10 to 15 mg/l; for perch (*Perca fluviatilis*) at levels of 60 mg/l; for roach (*Rutilus rutilis*) at levels of 40 to 60 mg/l; for water flea (*Daphnia* sp.) at levels of 2 to 4 mg/l; and for *Gammarus* sp. at levels of 4 mg/l. The damage was shown within three days for the various aquatic organisms tested. Damage also resulted if the fish were exposed to much lower concentrations for longer periods of time (Wilber 1969).[299]

Recommendation

Because of a chronic effect of long-term exposure of fish to thallium, tests should be conducted for at least 20 days on sensitive species. Techniques should measure circulatory disturbances (blood pressure) and other sublethal effects in order to determine harmful concentrations. The concentration in sea water should not exceed 0.05 of this concentration. On the basis of data available at this time, it is suggested that concentrations of thallium equal to or exceeding 0.1 mg/l constitute a hazard in the marine environment, and levels less than 0.05 mg/l present minimal risk of deleterious effects.

Uranium

Uranium is present in wastes from uranium mines and nuclear fuel processing plants, and the uranyl ion may naturally occur in drainage waters from uranium-bearing ore deposits. Small amounts may also arise from its use in tracer work, chemical processes, photography, painting and glazing porcelain, coloring glass, and in the hard steel of high tensile strength used for gun barrels.

Many of the salts of uranium are soluble in water, and it is present at about 3 μg/l in sea water. A significant proportion of the uranium in sea water is in the form of stable complexes with anionic constituents. It has been estimated that uranium has a residence time of 3×10^6 years in the oceans (Goldberg et al. 1971),[172] a span that makes it one of the elements with the slowest turnover time. Uranium is stabilized by hydrolysis which tends to protect it against chemical and physical interaction and thus prevents its removal from sea water. The salts are considered to be 4 times as germicidal as phenol to aquatic organisms.

Natural uranium (U-238) is concentrated from water by the algae *Ochromonas* by a factor of 330 in 48 hours (Morgan 1961).[232] Using River Havel water, Bringmann and Kuhn (1959a,[131] 1959b[132]) determined the threshold effect of uranyl nitrate, expressed as uranium, at 28 mg/l on a protozoan (*Microregma*), 1.7 to 2.2 mg/l on *Escherichia Coli*, 22 mg/l on the alga *Scenedesmus*, and 13 mg/l on *Daphnia*. Tarzwell and Henderson (1956)[277] found the sulfate, nitrate, and acetate salts of uranium considerably more toxic to fathead minnows (*Pimephales promelas*) on 96-hour exposure in soft water than in hard water, the 96-hr LC50 for uranyl sulfate being 2.8 mg/l in soft water and 135 mg/l in hard water.

The sparse data for uranium toxicity in sea water suggest that uranyl salts are less toxic to marine organisms than to freshwater organisms. Yeasts in the Black Sea were found to be more active than the bacteria in taking up uranium (Pshenin 1960).[251] Studies by Koenuma (1956)[205] showed that the formation of the fertilization membrane of *Urechis* eggs was inhibited by 250 mg/l of uranyl nitrate in sea water, and that this concentration led to polyspermy.

Recommendation

It is recommended that an application factor of 0.01 be applied to marine 96-hour LC50 data for

the appropriate organisms most sensitive to uranium. On the basis of data available at this time it is suggested that concentrations of uranium equal to or exceeding 0.5 mg/l constitute a hazard in the marine environment, and levels less than 0.1 mg/l present minimal risk of deleterious effects.

Vanadium

Vanadium occurs in various minerals, such as chileite and vanadinite. It is used in the manufacture of vanadium steel. Vanadates were used at one time to a small extent for medicinal purposes. Vanadium has been concentrated by certain marine organisms during the formation of oil-bearing strata in geological time. Consequently, vanadium enters the atmosphere through the combustion of fossil fuels, particularly oil. In addition, eighteen compounds of vanadium are used widely in commercial processes (Council on Environmental Quality 1971).[141]

Recommendation

It is recommended that the concentration of vanadium in sea water not exceed 0.05 of the 96-hour LC50 for the most sensitive species.

Zinc

Most of the available information on zinc toxicity is for freshwater organisms, and for this reason the reader is referred to the discussion of zinc in Section III, p. 182.

Recommendation

Because of the bioaccumulation of zinc through the food web, with high concentrations occurring particularly in the invertebrates, it is recommended that an application factor of 0.01 be applied to marine 96-hour LC50 data for the appropriate organisms most sensitive to zinc. On the basis of data available at this time, it is suggested that concentrations of zinc equal to or exceeding 0.1 mg/l constitute a hazard in the marine environment, and levels less than 0.02 mg/l present minimal risk of deleterious effects.

It should be noted that there is a synergistic effect when zinc is present with other heavy metals, e.g., Cu and Cd, in which case the application factor may have to be lowered by an order of magnitude (LaRoche 1972).[211]

OIL IN THE MARINE ENVIRONMENT

Oil is becoming one of the most widespread contaminants of the ocean. Blumer (1969)[319] has estimated that between 1 and 10 million metric tons of oil may be entering the oceans from all sources. Most of this influx takes place in coastal regions, but oil slicks and tar balls have also been observed on the high seas (Horn et al. 1970,[334] Morris 1971[343]). Collections of tar balls were made by towing a neuston net which skims the surface, and the investigators found that the tar balls were more abundant than the normal sargassum weed in the open Atlantic, and that their nets quickly became so coated with tar and oil that they were unusable. Thus, oil pollution of the sea has become a global problem of great, even though as yet inadequately assessed, significance to the fisheries of the world.

Sources of Oil Pollution

Although accidental oil spills are spectacular events and attract the most public attention, they constitute only about 10 per cent of the total amount of oil entering the marine environment. The other 90 per cent originates from the normal operation of oil-carrying tankers, other ships, offshore production, refinery operations, and the disposal of oil-waste materials (Table IV-4).

Two sources of oil contamination of the sea not listed in Table IV-4 are the seepage of oil from underwater oil reservoirs through natural causes and the transport of oil in the atmosphere from which it precipitates to the surface of the sea. Natural seepage is probably small compared to the direct input to the ocean (Blumer 1972);[320] but the atmospheric transport, which includes hydrocarbons that have evaporated or been emitted by engines after incomplete combustion, may be greater than the direct input.

Some of these sources of oil pollution can be controlled more rigorously than others, but without application of adequate controls wherever possible the amount of petroleum hydrocarbons entering the sea will increase. Our technology is based upon an expanding use of petroleum; and the production of oil from submarine reservoirs and the use of the sea to transport oil will both increase. It is estimated that the world production of crude oil in 1969 was nearly 2 billion tons; on this basis total losses to the sea are somewhat over 0.1 per cent of world production.

TABLE IV-4—Estimated Direct Petroleum Hydrocarbon Losses to the Marine Environment (Airborne Hydrocarbons Deposited on the Sea Surface are Not Included)
(Millions of tons)

	1969	1975 (estimate)[a]		1980 (estimate)	
		Min	Max	Min	Max
1. Tankers	.530	.056	.805	.075	1.062
2. Other ships	.500	.705	.705	.940	.940
3. Offshore production	.100	.160	.320	.230	.460
4. Refinery operations	.300	.200	.450	.440	.650
5. Oil wastes	.550	.825	.825	1.200	1.200
6. Accidental spills	.200	.300	.300	.440	.400
TOTAL	2.180	2.246	3.405	3.325	4.752
Total Crude Oil Production	1820	2700		4000	

[a] The minimum estimates assume full use of known technology; the maximums assume continuation of presen practices.

Revelle et al. 1972[345].

Some losses in the exploitation, transportation, and use of a natural resource are inevitable; but if this loss ratio cannot be radically improved, the oil pollution of the ocean will increase as our utilization increases.

Biological Effects of Petroleum Hydrocarbons

Description of Oil Pollution Oil is a mixture of many compounds, and there are conflicting views concerning its toxicity to marine organisms. Crude oils may contain thousands of compounds, and will differ markedly in their composition and in such physical properties as specific gravity, viscosity, and boiling-point distribution. The hydrocarbons in oil cover a wide range of molecular weights from 16 (methane) to over 20,000. Structurally, they include aliphatic compounds with straight and branched chains, olefins, and the aromatic ring compounds. Crude oils differ mainly in the relative concentrations of the individual members of these series of compounds. The various refinery processes to which oil is subjected are designed to isolate specific parts of the broad spectrum of crude oil compounds, but the refined products themselves remain complex mixtures of many types of hydrocarbons.

In spite of the many differences among them, crude oils and their refined products all contain compounds that are toxic to species of marine organisms. When released to the marine environment, these compounds react differently. Some are soluble in the water; others evaporate from the sea surface, form extensive oil slicks, or settle to the bottom if sand becomes incorporated in the oil globule. More complete understanding of toxicity and the ecological effects of oil spills will require studies of the effects of individual components, or at least of classes of components, of the complex mixture that made up the original oil. The recent development of gas chromatography has made it possible to isolate and identify various fractions of oil and to follow their entry into the marine system and their transfer from organism to organism.

An oil slick on the sea surface can be visually detected by iridescence or color, the first trace of which is formed when 100 gallons of oil spread over 1 square mile (146 liters/km^2) (American Petroleum Institute 1949).[317] The average thickness of such a film is 0.145 microns. Under ideal laboratory conditions, a film 0.038 microns thick can be detected visually (American Petroleum Institute 1963).[318] For remote sensing purposes, oil films with a thickness of 100 microns can be detected using dual polarized radiometers, 1 micron using radar imagery, and 0.1 microns using multispectral imagery in the UV region (Catoe and Orthlieb 1971).[323] A summary of remote sensing capabilities is presented in Table IV-5. Because remote sensing is less effective than the eye in detecting surface oil, any concentration of oil detectable by remote means currently available will exceed the recommendations given below.

The death of marine birds from oiling is one of the earliest and most obvious effects of oil slicks on the sea surface. Thousands of seabirds of all varieties are often involved in a large spill. Even when the birds are cleaned, they frequently die because the toxic oil is ingested in preening their feathers. Dead oiled birds are often found along the coast when no known major oil spill has occurred, and the cause of death remains unknown.

When an oil spill occurs near shore or an oil slick is brought to the intertidal zone and beaches, extensive mortality of marine organisms occurs. When the Tampico Maru ran aground off Baja California in 1957, about 60,000 barrels of spilled diesel fuel caused widespread death among lobsters, abalones, sea urchins, starfish, mussels, clams, and hosts of smaller forms (North 1967).[344] A beneficial side effect of this accident was also noted by North. When the sea urchins that grazed on the economically important kelp beds of the area were killed in massive numbers by the oil spill, huge canopies of kelp returned within a few months (see p. 237). The oil spills from the wreck of the tanker Torrey Canyon and the Santa Barbara oil well blowout both involved crude oil, and in both cases oil reached the beaches in variable amounts some time after release. The oil may thus have been diluted and modified by evaporation or sinking before it reached the beach. In the Santa Barbara spill many birds died, and entire plant and animal communities in the intertidal zone were killed by a layer of encrusting oil often 1 or 2 centimeters thick (Holmes 1967).[333] At locations where the oil film was not so obvious, intertidal organisms were not severely damaged (Foster et al. 1970).[327] In the case of the Torrey Canyon, the deleterious effects have been attributed more to the detergents and dispersants used to control the oil than to the oil itself (Smith 1968).[347]

A relatively small oil spill in West Falmouth, Massachusetts, occurred within a few miles of the Woods Hole Oceanographic Institution in September 1969. An oil barge, the Florida, was driven onto the Buzzards Bay Shore where it released between 650 and 700 tons of No. 2 fuel oil into the coastal waters. Studies of the biological and chemical effects of this spill are continuing, more than two years after the event (Blumer 1969,[319] Hampson and Sanders 1969,[331] Blumer et al. 1970,[322] Blumer and Sass 1972[321]). Massive destruction of a wide range of fish, shellfish, worms, crabs, other crustaceans, and invertebrates occurred in the region immediately after the accident. Bottom-living fish and lobsters were killed and washed ashore. Dredge samples taken in 10 feet of water soon after the spill showed that 95 per cent of the animals recovered were dead and the others moribund. Much of the evidence of this immediate toxicity disappeared within a few days, either because of the breaking up of the soft parts of the organism, burial in the sediments, or dispersal by water currents. Careful chemical and biological analyses reveal, however, that not only has the damaged area been slow to recover but the extent of the damage has been expanding with time. A year and a half after the spill, identifiable

TABLE IV-5—Summary of Remote Sensor Characteristics For Oil Detection

Wave length	Detection mechanism	Performance summary	Possible sensor configuration					Comments
			Type	Resolution	Weight	Volume	Swath width	
Ultraviolet ($\leq 0.4 \mu m$).	Reflectance differential (Oil/Water contrast)	Reflective signature a. Repeatable positive response from thin slicks ($\sim .1$ micron). b. Variable response from thicker slicks dependent upon oil type, water quality and illumination conditions. c. Atmospheric haze limitations major. d. Signal limitations prevent night-time detection.	UV Vidicon	500 lines/frame (high scene illumination) 100–200 lines/frame (low scene illumination)	33 lbs.	2 cu. ft.	40° FOV (727 ft @ 10 K)	Developed equipment available for UV vidicon and/or scanner. Integrates well with CRT display.
	Fluorescence	Fluorescence signature 1. Artificial Excitation (narrow-band) a. Spectral character strongly correlated to oil thickness. b. Intensity strongly correlated to oil type (API) and oil thickness, weakly correlated to temperature. c. Decay characteristics moderately to strongly correlated to oil type, uncorrelated to oil thickness. d. All characteristics independent of ambient illumination conditions. 2. Solar excitation (broad-band) a. Spectral character moderately to weakly correlated to oil type and thickness. b. Intensity strongly correlated to oil type, oil thickness and ambient illumination conditions. c. Decay characteristics not detectable. d. Signal limitations prevent operation except under strong solar illumination.	UV Scanner	2 mr	90 lbs.	3.5 cu. ft.	2.7 mi @ 10 K	Line scanner may require data buffer for high resolution, real time display, or film processor
			Pulsed Laser	1 mr	150 lbs.	4 cu. ft.	10 ft. @ 10 K	Effective against thin and thick slicks under solar, or artificial illumination. Active laser system sensitivity limitations hinder use in detection or mapping mode. Identification capability very good, with moderate to good thickness determination.
Visible (0.4 to .7 μm)	Reflectance Differential (0.1/Water Contrast)	Reflective Signature a. Variable response from all slicks dependent upon thickness, oil type, water quality and illumination conditions	Aerial Cameras RC-8	2 ft. @ 10 K	190 lbs.	11.76 cu. ft.	74° FOV 3.5 mi. @ 10 K	Aerial cameras real time display not possible.
		b. Signal limitations prevent moonless nighttime detection.	500-EL	3.5 ft. @ 10 K	16 lbs.	.4 cu. ft		Sensitivity limitations prevent night-time operations.
		c. False alarm problem significant. d. Atmospheric haze limitation major.	KA-62	" "	61.5 lbs.	5.24 cu. ft.		Compensation for atmospheric haze difficult.
		e. Maximum contrast between oil and water occurs at (.38 to .45 μm) and (.6 to .68 μm). f. Minimum contrast between oil and water occurs at (.45 to .58 μm) g. Best contrast achieved with overcast sky.	Vidicon	500 lines/frame	33 lbs.	2 cu. ft.	40° FOV with zoom lens 7270 ft. at 10 K	UV photography great potential for detecting oil. Color is good; however, sunlight gives false response. Panchromatic, IR and color photography and TV give good results only when oil is thick and ropy. Vidicon useful for real-time detection and mapping at various wave lengths, giving option for good detection with negligible false alarms for day operation and fair-to-good detection with low false alarms for night operation. Display characteristics optimum for surveillance.
Infrared Near Infrared (0.6 to 0.1 μm)	Reflectance Differential (0.1/Water Contrast)	Reflective Signature a. Repeatable positive response from all slicks under all conditions.	Line Scanner	2 mr	90 lbs.	4.8 cu. ft.	2.7 mi @ 10 K	Line scanner oil-slick response variable but essentially predictable, but may have some false alarm problems.
Far Infrared (8 to 14 μm)	Thermal Emission Differential	b. Moonless night-time detection capability. c. False alarm problems negligible. d. Atmospheric haze limitation moderate.	Framing Scanner	4 mr	220 lbs	3.5 cu. ft.	25° FOV	

TABLE IV-5—Summary of Remote Sensor Characteristics For Oil Detection—Continued

Wave length	Detection mechanism	Performance summary	Possible sensor configuration					Comments
			Type	Resolution	Weight	Volume	Swath width	
		Thermal Signature a. Variable response dependent grossly upon oil type and dependent significantly upon thickness and solar heating. Variability predictable to significant degree (slicks >10 μm) b. Day night detection independent of illumination conditions. c. False alarm problem slight. d. Atmospheric haze limitations moderate to slight.						Day night detection under VFR conditions. Real time display capabilities good but limited to "single-look" display generation. Developed equipment available.
Microwave	Emissive Differential (Oil Water Contrast) Wave Structure Modification	Emissive Signature a. Emissivity of petroleum products is significantly higher than that of a calm sea surface. b. Crude oil pollutants have decreasing dielectric constants (increasing emissivity) with increasing API gravity. c. Microwave signature of oil film inversely proportional to sensor wave length. d. The horizontal polarized microwave signature of oil is twice the vertically polarized signature of an oil slick on a flat water surface. e. Detection improves with decreasing sensor wave lengths and becomes poorer as the sea state increases. f. Atmospheric cloud limitations moderate to slight. g. Can effectively detect slicks less than 0.1 mm at viewing. h. Dual frequency microwave techniques show great promise in measuring oil slick thickness.	Line Scanning Imager	1.4°	68 lbs.	3 cu. ft.	2.7 mi @ 10 K	Clouds that are raining between sensor and slick as well as very high sea states hamper performance. Technology for equipment development available. Real time display consists of facsimile and/or CRT.
Radar	Wave Structure Modification Scattering Cross-section Differential	Reflective Signature a. Oil film on surface of water suppresses capillary which results in a significant difference in energy back scattered from contaminated surface and that scattered from surrounding clean water (from oil slicks very little energy back scattered by three orders of magnitude). b. Vertical polarization capable of detecting and mapping oil slick less than 1 micron. c. Atmospheric cloud limitations slight.	Forward Scanning (35 GHz)	100×100 ft.²	~600 lbs.	10 cu. ft.	38 mi @ 12 K	Technology exists for equipment development of forward scanning and synthetic aperature radar.
			Synthetic Aperature (3.3 GHz)	100×100 ft.²	~1500 lbs	17 cu. ft.	150 mi @ 36 K	Real time display possible for forward scanning radar via facsimile and/or CRT; synthetic aperature radar requires optical processing.

fractions of the source oil were found in organisms that still survived on the perimeter of the area. Hydrocarbons ingested by marine organisms may pass through the wall of the gut and become part of the lipid pool (Blumer et al. 1970).[322] When dissolved within the fatty tissues of the organisms, even relatively unstable hydrocarbons are preserved. They are protected from bacterial attack and can be transferred from food organism to predators and possibly to man.

The catastrophic ecological effects of the oil spills of the Tampico Maru, and the Florida appear to be more severe than those reported from other oil spills such as the Torrey Canyon and the Santa Barbara blowout. The Tampico Maru and the Florida accidents both released refined oils (in one case diesel oil and in the other, No. 2 fuel oil) and both occurred closer to shore than either the Torrey Canyon or the Santa Barbara accidents which released crude oil. The differences in the character of the oil and the proximity to shore may account for the more dramatic effects of the first two accidents, but it is clear that any release of oil in the marine environment carries a threat of destruction and constitutes a danger to world fisheries.

Persistence of Oil in the Ocean As mentioned above, oil can be ingested by marine organisms and incorporated in their lipid pool. Hydrocarbons in the sea are also degraded by marine microorganisms. Very little is known as yet about the rate of this degradation, but it is known that no single microbial species will degrade any whole crude oil. Bacteria are highly specific, and several species will probably be necessary to decompose the numerous types of hydrocarbons in a crude oil. In the process of decomposition, intermediate products will be formed and different species of bacteria and other microorganisms may be required to attack these decomposition products (ZoBell 1969).[318]

The oxygen requirement of microbial oil decomposition is severe. The complete oxidation of one gallon of crude oil requires all the dissolved oxygen in 320,000 gallons of air-saturated sea water (ZoBell 1969).[318] It is clear that oxidation might be slow in an area where previous pollution has depleted the oxygen content. Even when decomposition of oil proceeds rapidly, the depletion of the oxygen content of the water by the microorganisms degrading the oil may have secondary deleterious ecological effects. Unfortunately, the most readily attacked fraction of crude oil is the least toxic, i.e., the normal paraffins. The more toxic aromatic hydrocarbons, especially, the carcinogenic polynuclear aromatics, are not rapidly degraded.

That our coastal waters are not devoid of marine life, after decades of contamination with oil, indicates that the sea is capable of recovery from this pollution. However, increasing stress is being placed on the estuarine and coastal environment because of more frequent oil pollution incidents near shore; and once the recovery capacity of an environment is exceeded, deterioration may be rapid and catastrophic. It is not known how much oil pollution the ocean can accept and recover from, or whether the present rate of addition approaches the limit of the natural system.

It appears that the oceans have recovered from the oil spilled during the six years of the second World War, though some unexplained recent oil slicks have been attributed to the slow corrosion of ships sunk during that conflict. It has been estimated (SCEP)[315] that during the war, the United States lost 398 vessels with a total oil capacity of about 1 million tons, and that another 3 million tons of oil were lost through the sinking of ships of other combatants during the same period. These losses were large in the context of the 1940's, but the total for that period was only about twice the annual direct influx to the ocean at the present time. Although no extensive deleterious effects of these sinkings and oil releases on the fisheries catch of the world have been found, it must be emphasized again that when a pollutant is increasing yearly in magnitude past history is not a reliable source of prediction of future effects.

The Toxicity of Oil There is a dearth of dependable observations on the toxicity of oil to marine organisms. It is difficult to evaluate the toxicity of this complex mixture of compounds which is not miscible with sea water. A variety of techniques have been used which are not intercomparable. In some experiments, oil is floated on the water in the test container, and the concentration given is derived from the total quantity of oil and the total quantity of water. This is clearly not the concentration to which the organism has been exposed. In other experiments, extracts of oil with hot water or with various solvents have been added to the test jar without identification of the oil fraction being tested. In still other cases, care has been taken to produce a fine emulsion of oil in sea water more representative of the actual concentration to which the test organism is exposed. Considering the differences in the meaning of "concentration" in these tests and the variation in sensitivity of the test organisms, it is not surprising that the ranges of toxicity that can be found in the literature vary by several orders of magnitude.

Studies of the biological effects of oil have been reviewed by Clark (1971).[325] Mironov (1971)[342] carried out toxicity studies by comparable techniques using a variety of marine organisms. In testing eleven species of phytoplankton, he found that cell division was delayed or inhibited by concentrations of crude oil (unspecified type) ranging from 0.01 to 1000 ppm. He also showed that some copepods were sensitive to a 1 ppm suspension of fresh or weathered crude oil and of diesel oil. Freegarde et al. (1970)[328] found that the larvae of *Ballanus ballanoides* and adult *Calanus* copepods maintained in a suspension of crude oil ingest, without apparent harm, droplets of oil that later appear in the feces. Mironov (1967)[341] found 100 per cent mortality of developing flounder spawn at concentrations of three types of oil ranging from 1 to 100 ppm and an increased abnormality of development at longer periods of time in concentrations as low as 0.01 ppm. In contrast other experimenters have found that concentrations of several per cent are necessary to kill adult fish in a period of a few days (Chipman and Galtsoff 1949,[324] Griffith 1970[329]).

The evidence is clearer that a combination of oil and detergents is more toxic than oil alone. This was first definitely established in studies of the Torrey Canyon spill (Smith 1968),[347] and the toxicity of the various detergents used in this operation is discussed by Corner et al. (1968).[326] The four detergents tested were all more toxic than Kuwait crude oil, and all showed signs of toxicity between 2 and 10 ppm. The solvents used with these detergents were also highly toxic but tended to lose their toxicity over time through evaporation. A bioassay test carried out by the Michigan Department of Natural Resources (1969)[338] revealed that the least toxic detergent mixed with oil could be a hundred times as concentrated (1800 ppm) as the most toxic (14 ppm) and cause the same toxic effect. La Roche et al. (1970)[337] defined bioassay procedures for oil and oil dispersant toxicity evaluation using fish, *Fundulus heteroclitus*, and the sandworm, *Nereis virens* (Table IV-6).

The mortality of seabirds as a result of oil pollution is direct and immediate, and in a major oil spill, is measured in the thousands. The diving birds which spend most of their life at sea are most prone to death from oil pollution, but any bird that feeds from the sea or settles on it is vulnerable. In oil-matted plumage air is replaced by water

TABLE IV-6—Determinations (Summarized) of Acute Toxicities of 10 Chemical Dispersants Alone and in Combination with Crude Oil to Sandworm* (Nereis virens) *and Mummichog* (Fundulus heteroclitus) *in Laboratory Bioassay Tests

Substance	96 hour LC50 (ml/l)	
	Nereis	Fundulus
Crude oil A		16.5
Crude oil B	6.1	8.2
Oil and dispersants[a]	.055-.781	.187-1
Dispersants	.007-7.10	.008-2

[a] Ranges of values for 10 dispersants mixed 1 part dispersant to 10 parts of oil by volume.
LaRoche et al. 1970[337].

causing loss of both insulation and buoyancy, and oil ingested during preening can have toxic effects.

Hartung and Hunt (1966)[332] fed oils directly to birds by stomach tube and later analyzed the pathological and physiological effects through autopsies. The lethal dose for three types of oil ranged from 1 ml to 4 ml per kilogram (ml/kg) when the birds were kept outdoors under environmental stress. The experimenters concluded that a duck could typically acquire a coating of 7 grams of oil and would be expected to preen approximately 50 per cent of the polluting oil from its feathers within the first few days. Enough of this could easily be ingested to meet the lethal dosage of 1 to 4 ml/kg. Thus, birds that do not die promptly from exposure to cold or by drowning as a result of oil pollution may succumb later from the effects of ingestion.

Corrective Measures

The only effective measure for control of oil pollution in the marine environment is prevention of all spills and releases. The time-lag involved in corrective methods means that some damage will inevitably occur before the corrective measures take effect. Furthermore, the soluble parts of the oil already in the water will not be removed by any of the present methods of post-spill cleanup.

Control measures have been introduced that appreciably reduce excessive oil pollution from normal tanker operations (see Table IV-4). The load on top (LOT) process concentrates waste oil that is ultimately discharged with the new cargo (IMCO 1965a,[335] 1965b[336]). This procedure recovers somewhat more than 98 per cent of oil that would otherwise be released to the sea. It has been estimated (Revelle et al. 1972)[345] that 80 per cent of the world fleet uses these control measures today, and if they continue to do so faithfully these ships will contribute only 3.0×10^4 tons of the total tonnage of oil loss. In contrast, the 20 per cent of the fleet not using these control measures contributes 5×10^6 tons. If these control measures were not in use by a major fraction of the tanker fleet, the contamination of the sea from this source would be about five times greater than it is today.

Among the earliest methods for the cleanup of spilled oil was to pick up or bury the material that came ashore while disregarding the oil that remained at sea. It was found that the use of straw to absorb the oil made this cleanup procedure easier, and in the cleanup of the Arrow oil spill (Ministry of Transport, Canada 1970),[340] peat moss was found to be an effective absorbent for Bunker C oil. Recent studies promise mechanical means for handling and cleaning sand contaminated with oil by use of earth moving equipment, fluid-bed, and froth flotation techniques (Gumtz and Meloy 1971,[330] Mikolaj and Curran 1971,[339] Sartor and Foget 1971).[346]

The use of detergents to treat oil slicks is essentially cosmetic. It removes the obvious evidence of oil and for that reason appeals to the polluter. However, after treatment with detergent, the oil is dispersed in the form of fine droplets and becomes even more available to the biota of the sea than it would be if it were left in the form of a surface film. Because of the finer degree of dispersion, the soluble toxic fractions dissolve more rapidly and reach higher concentrations in sea water than would result from natural dispersal. The droplets themselves may be ingested by filter-feeding organisms and thus become an integral part of the marine food chain. Some of the oil may pass through the gut in the feces of these organisms, but Blumer et al. (1970)[322] have shown that it can pass through the gut wall and be incorporated in the organism's lipid pool. It can thus be transferred from organism to organism and, potentially, into the food that man takes from the ocean for his use.

Sinking of oil has been achieved by scattering talc or chalk on the oil causing it to agglutinate into globules of greater density than sea water. Such sunken oil tends to kill bottom fauna before even the motile bottom dwellers have time to move away. The sessile forms of commercial importance, such as clams, oysters and scallops, cannot escape, and other motile organisms such as lobsters (*Homarus americanus*) may actually be attracted in the direction of the spill where exposure will contaminate or kill them. Little is known about the rate of degradation of oil in bottom sediments, but it is known that some fractions will persist for over two years (Blumer 1969,[319] Blumer and Sass 1972[321]). Chipman and Galtsoff (1949)[324] showed that the toxicity of oil is not diminished by adsorption on carbonized sand which can be used as a sinking agent.

Efforts were made to burn the oil in both the Torrey Canyon and the Wafra, which was wrecked off the coast of South Africa in 1971. When oxidation is complete, oil is converted to carbon dioxide and water and removed as a pollutant. Burning oil within a tanker, however, is difficult; and it has not been successful even when oxidants are added. Volatile fractions may burn off quickly, but most of the oil resists combustion. Incomplete combustion is therefore not only more common, but the smoke and volatile oils themselves become atmospheric pollutants many of which ultimately return to the sea through precipitation and accumulation on the water surface. Oil can be burned on the surface of the sea by using wicks or small glass beads to which the oil clings thus removing itself from the quenching effects of the water. The use of "seabeads" was successful in burning Bunker C oil on the beach and moderately successful in burning a slick in two to three foot seas in the cleanup following the wreck of the Arrow (Ministry of Transport, Canada 1970).[340] However, during burning, the elevated temperature of the oil increases the solubility in water of the most toxic components, and this can cause greater biological damage than if the oil is left unburned.

Mechanical containment and removal of oil appear to be ideal from the point of view of avoiding long-term biological damage, but however promptly such measures are taken, some of the soluble components of the oil will enter the water and it will not be possible to remove them. A variety of mechanisms for containing oil have been proposed, such as booms with skirts extending into the water. Various surface skimmers to collect oil and pump it into a standby tanker have been conceived. Unfortunately, most wrecks occur during less than ideal weather conditions which makes delivery and deployment of mechanical devices difficult. Floating booms are ineffective in a rough sea, because even if they remain properly deployed, oil can be carried over the top of them by wind and splashing waves or under them by currents. In protected waters, however, recovery can be quite effective, and among the methods of oil removal used today, booms are one of the most effective if conditions for their use are favorable.

Microbiological degradation is the ultimate fate of all oil left in the sea, but as was mentioned previously, the oxygen requirement for this is severe. There is also the problem of providing other nutrients, such as nitrogen and phosphorus, for the degrading bacteria. Nevertheless, this process is a "natural" one, and research into increasing the rate of bacteriological degradation without undesirable side effects is to be encouraged.

Although an ultimate solution to the cleanup of oil spills is desperately needed, prevention of spills remains the most effective measure. When wrecks occur, every effort should be made to offload the oil before it enters the marine environment. Oil spills that occur in harbors during transfer of oil to a refinery or of refined oil to a tanker should be more easily controlled. Portable booms could confine any oil released and make possible recovery of most harbor spillage. Available technology is adequate to prevent most accidental spills from offshore well drilling or operations. It is necessary to require that such technology be faithfully employed.

Recommendations

No oil or petroleum products should be discharged into estuarine or coastal waters that:

- **can be detected as a visible film, sheen, or discoloration of the surface, or by odor;**
- **can cause tainting of fish or edible invertebrates or damage to the biota;**
- **can form an oil deposit on the shores or bottom of the receiving body of water.**

In this context, discharge of oil is meant to include accidental releases that could have been prevented by technically feasible controls.

Accidental releases of oil to the marine environment should be reclaimed or treated as expeditiously as possible using procedures at least equivalent to those provided in The National Contingency Plan of 1970. The following recommendations should be followed to minimize damage to the marine biota.

- **Oil on the sea surface should be contained by booms and recovered by the use of surface skimmers or similar techniques.**
- **In the event of a tanker wreck, the oil remaining in the hulk should be off-loaded.**
- **Oil on beaches should be mechanically removed using straw, peat moss, other highly absorbent material, or other appropriate techniques that will produce minimal deleterious effects on the biota.**
- **Failing recovery of oil from the sea surface or from a wrecked tanker, efforts should be made to burn it in place, provided the contamination is at a safe distance from shore facilities. If successful, this will minimize damage to the marine biota.**
- **Dispersants should be used only when necessary and should be of minimal potential toxicity to avoid even greater hazard to the environment.**
- **Sinking of oil is not recommended.**

All vessels using U.S. port facilities for the purpose of transporting oil or petroleum products should be required to demonstrate that effective procedures or devices, at least equivalent to the "Load on Top" procedure, are used to minimize oil releases associated with tank cleaning.

In order to protect marine wildlife:

- **recommendations listed above should be followed;**
- **a monitoring program should follow long-term trends in petroleum tar accumulation in selected areas of the oceans;**
- **no oil exploration or drilling should be permitted within existing or proposed sanctuaries, parks, reserves or other protected areas, or in their contiguous waters, in a manner which may deleteriously affect their biota;**

- **oil exploration or drilling should not be conducted in a manner which may deleteriously affect species subject to interstate or international agreements.**

TOXIC ORGANICS

The toxic organics constitute a considerable variety of chemical compounds, almost all of which are synthetic. The total production of synthetic organic chemicals in the U.S. in 1968 was 120,000 million pounds, a 15 per cent increase over 1967; 135,000 million pounds were produced in 1969, a 12 per cent increase over 1968 (United States Tariff Commission 1970).[377] This figure, in the order of 5×10^{13} grams, may be compared with the total productivity of the sea, which is in the order of 2×10^{16} grams of carbon incorporated into phytoplankton per year (Ryther 1969).[373] When considered in a global and future context, the production of synthetic chemicals by man cannot be considered an insignificant fraction of nature's productivity.

The majority of the synthetic organic chemicals, including those considered toxic, are readily degradable to elementary materials which reenter the chemical cycles in the biosphere. These pose no long-term hazard if applied or released into the environment in quantities sufficiently small to meet the recommendations for mixing zones (see p. 231).

The chemicals of most concern are the more stable compounds that enter the environment, whether they are introduced incidentally as waste materials or deliberately through their use. The toxicity, chemical stability, and resistance to biological degradation of such chemicals are factors that must be considered in assessing their potential effects on ecosystems. Moreover, because of the partitioning of nonpolar compounds among the components of marine ecosystems, relatively high concentrations of these, including halogenated hydrocarbons, are frequently found in organisms.

Only recently it was discovered that polychlorinated biphenyls (PCB), a class of chlorinated hydrocarbons used in a variety of industrial applications, were widespread contaminants in marine ecosystems (Duke et al.).[354a] Concentrations up to or higher than 1000 ppm in the body fat of estuarine birds have been recorded in both Europe and North America (Risebrough et al. 1968,[371] Jensen et al. 1969[360]). Moreover, both DDT and PCB have been found in organisms from depths of 3200 meters in the open North Atlantic Ocean (Harvey et al. 1972).[359]

The discovery of a man-made contaminant such as PCB, unknown in the environment a few years ago, in such unexpectedly high concentrations in marine organisms raises several questions. Are the concentrations of these compounds still increasing in the marine environment and at what rate, and what are the long-term effects upon the marine communities? Is it possible that other pollutants, undetected by the methodologies that measure the chlorinated hydrocarbons, are present in comparable amounts?

Criteria employed in the past to protect freshwater ecosystems were based on data now seen to be inadequate and on an approach that looked at pollutant concentrations in waste water effluents rather than in the receiving system. Evidently it is necessary to attempt to relate the amounts of input into the ecosystem to the levels in the various components of the ecosystem, including indicator organisms. The concentrations of a persistent pollutant in an indicator organism are considered the best way of following accumulation trends in an aqueous ecosystem that serves as a sink for the pollutant, once the capacity of the ecosystem to absorb the pollutant has been determined. If the concentrations in the indicator organisms exceed those considered safe for the ecosystem, input should then be reduced, restricted, or eliminated until environmental levels are acceptable on the basis of established criteria. Inputs of persistent pollutants into the marine environment, however, are in many cases indirect and not immediately controllable, e.g. river runoffs, atmospheric fallout, and dumping by foreign and domestic ships. The sources of the chemicals in atmospheric fallout may be located anywhere in the world.

Different recommendations must therefore be developed to protect the marine environment from increasing amounts and varieties of organic pollutants that might be anticipated over the next century. The same recommendations may be applied to estuaries, but these must also be protected from a variety of chemicals that are less persistent and pose no long-term hazard, but that may, because of toxic effects upon organisms, cause unacceptable amounts of damage. These include many of the pesticides, components of sewage, biological wastes from slaughter houses, and other organic wastes from industry.

Acute toxicity values and subacute effects of pesticides on marine life are listed in Appendix III–Table 6, and in Table IV-7, p. 265. Table IV-7 is a summary of the "most sensitive" organisms taken from Appendix III–Table 6 and includes a list of chemicals that are considered to have potential environmental importance in estuarine or marine ecosystems. The list includes many of the pesticides that are readily degradable in the environment but because of their high toxicity are potentially dangerous to estuarine ecosystems. The list, which should be revised as new data become available, proposes a minimum number of such chemicals. Appendix III–Table 6 includes the following information relative to the potential importance of each material as coastal and marine contaminants. (a) Production figures, which are taken from the 1969 Tariff Commission reports, are listed in the second column. The production figures provide a useful clue to the compounds that are of potential importance as marine pollutants. The order of the chemicals generally follows that of the Tariff Commission reports and is not intended to be a ranking in order of importance. (b) The third column of the table indicates

TABLE IV-7—Presence and Toxicity of Organic Chemicals in the Marine System

Chemical (1)	U.S. production pounds, gal./yr (2)	Presence in sea water or marine organisms (3)	Trophic accumulation (4)	Most sensitive organisms tested (5)	Formulation (6)	Conc. (ppb active ingredient in water) (7)	Method of assessment (8)	Test procedure (9)	Reference (10)
PESTICIDES, Total	1.1×10^9 lb	...	...	...					
Fungicides									
Fungicides, total	1.4×10^8	...	...	...					
Pentachlorophenol	4.6×10^7	Expected	Unknown	Insufficient data for marine organisms					
2,4,5-Trichlorophenol	Not available (1969) 2.8×10^7 (1968)	Unknown	Unknown	Crassostrea virginica American oyster	....	600	TLM	48 hr static lab bioassay	Davis and Hidu 1969[254]
Nabam (Ethylene bis[dithiocarbamic acid], disodium salt)	1.9×10^6	Unlikely	Unlikely	Dunaliella tertiolecta		100	.270. O. D. expt/O.D. control	10 day growth test	Ukeles 1962[876]
Hexachlorobenzene	Not available	Expected	Detected in birds (Vos et al., 1968)[878] Koeman and Genderen, 1970)[813]	Insufficient data for marine organisms					
Herbicides									
Herbicides, total	3.9×10^8	...	...	...					
Amitrole (3-amino-1,2,4-triazole)	Not available	Unlikely	Unlikely	Insufficient data					
Chloramben (3-amino-2,5-dichlorobenzoic acid, sodium salt)	Not available	Unlikely	Unlikely	Chlorococcum sp Phaeodactylum tricornutum	Methyl ester	2.5×10^3	50% decrease in growth	Growth measured as ABS. (525 mu) after 10 days	Walsh 1972[879]
Picloram (4-amino-3,5,6-trichloropicolinic acid) (Tordon™)	Not available	Unlikely	Unlikely	Isochrysis galbana		1×10^5	50% decrease in O_2 evolution*	....	Walsh 1972[879]
						5×10^4	50% decrease in growth	Measured as ABS. (525 mu) after 10 dys	Walsh 1972[879]
Simazine [2-chloro-4,6-bis-(ethylamino)-s-triazine]	Not available	Unlikely	Unlikely	Isochrysis galbana	Technical acid	500	50% decrease in growth	Measured as ABS. (525 mu) after 10 days	Walsh 1972[879]
				Phaeodactylum tricornutum	Technical aid	500	50% decrease in growth	Measured as ABS. (525 mu) after 10 days	Walsh 1972[879]
Atrazine [2-chloro-4-ethylamino-6-isopropyl-amino-s-triazine]	Not available	Unlikely	Unlikely	Chlorococcum sp., Chlamydomonas sp., Monochrysis lutheri	Technical acid	100	50% decrease in growth	Measured as ABS. (525 mu) after 10 days	Walsh 1972[879]
				Isochrysis galbana	Technical acid	100	50% decrease in O_2 evolution*	....	Walsh 1972[879]
				Phaeodactylum tricornutum	Technical acid	100	50% decrease in O_2 evolution*	....	Walsh 1972[879]
Monuron [3-(p chlorophenyl)-1,1-dimethylurea]	Not available	Unlikely	Unlikely	Protococcus sp.		20	.00 OPT. DEN. expt opt DEN control	10 day growth test	Ukeles 1962[876]
				Dunaliella tertiolecta		20	.00 OPT. DEN. expt opt DEN control	10 day growth test	Walsh 1972[879]
				Phaeodactylum tricornutum		20	.00 OPT. DEN. expt opt DEN control	10 day growth test	Ukeles 1962[876]
Diuron [3-(3,4-dichlorophenyl)-1,1-dimethylurea]	Not available	Unlikely	Unlikely	Protococcus		0.02	.52 OPT. DEN. expt opt DEN control	10 day growth test	Ukeles 1962[876]
				Monochrysis lutheri		0.02	.00 OPT. DEN. expt opt DEN control	10 day growth test	Ukeles 1962[876]
Maleic hydrazide [1,2-dihydropyridazine-3,6-dione]	2.8×10^6 lb.	Unlikely	Unlikely	Insufficient data					
Fenuron [1,1-dimethyl-3-phenylurea]	Not available	Unlikely	Unlikely	Chlorococcum sp.	Technical acid	750	50% decrease in growth	10 day growth test	Walsh 1972[879]
				Isochrysis galbana	Technical acid	750	50% decrease in growth	10 day growth test	Walsh 1972[879]
				Monochrysis lutheri	....	290	.67 OPT. DEN. expt/opt DEN control	10 day growth test	Ukeles 1962[876]
Ametryne [2-ethylamino-4-isopropylamino-6-methylmercapto-s-triazine]	Not available	Unlikely	Unlikely	Chlorococcum sp.	Technical acid	10	50% decrease in growth	Measured as ABS. (525 mu) after 10 days	Walsh 1972[879]
				Isochrysis galbana	Technical acid	10	50% decrease in O_2 evolution*	....	Walsh 1972[879]
				Monochrysis lutheri Phaeodactylum tricornutum	Technical acid	10	50% decrease in O_2 evolution*	....	Walsh 1972[879]

* /O_2 evolution measured by Gilson differential respirometer on 4 ml of culture in log phase. Length of test 90 minutes.

TABLE IV-7—*Presence and Toxicity of Organic Chemicals in the Marine System*—Continued

Chemical (1)	U.S. production pounds, gal./yr (2)	Presence in sea water or marine organisms (3)	Trophic accumulation (4)	Most sensitive organisms tested (5)	Formulation (6)	Conc. (ppb active ingredient in water) (7)	Method of assessment (8)	Test procedure (9)	Reference (10)
Herbicides, cont.									
Endothal [7-oxabicyclo-(2.2.1) heptane-2,3-dicarboxylic acid, disodium salt]	Not available	Unlikely	Unlikely	Mercenaria mercenaria Hard clam	....	1.25×10^{4}	TLM	12 day static lab bioassay	Davis and Hidu 1969[354]
MCPA [4-chloro-2-methyl-phenoxyacetic acid]	Not available	Unlikely	Unlikely	Crassostrea virginica American oyster	......	1.56×10^{4}	TLM	48 hr static lab bioassay	Davis and Hidu 1969[354]
2,4-D & derivatives	1.0×10^{8} lb	Unknown	Unknown	Crassostrea virginica American oyster	Ester	740	TLM	14 day static lab bioassay	Davis and Hidu 1969[354]
2,4,5-T & derivatives [2,4,5-trichlorophenoxy-acetic acid]	2.8×10^{7} lb	Unknown	Unknown	Dunaliella tertiolecta Isochrysis galbana	Technical acid	5×10^{4}	50% decrease in O_2 evolution*	....	Walsh 1972[379]
				Phaeodactylum tricornutum	Technical acid	5×10^{4}	50% decrease in growth	Measured as ABS. (525 mu) after 10 days	Walsh 1972[379]
Silvex [2-(2,4,5-trichloro-phenoxy)propionic acid]	1.6×10^{6}	Unlikely	Unlikely	Crassostrea virginica American oyster Dunaliella tertiolecta	....	710	TLM	14 day static lab bioassay	Davis and Hidu 1969[354]
Diquat [6,7-Dihydrodipyrido (1,2-a:2′,1′-c)pyrazidi-inium dibromide	Not available	Unlikely	Unlikely	Chlorococcum sp.	Dibromide	5×10^{6}	50% decrease in O_2 evolution*	....	Walsh 1972[379]
				Dunaliella tertiolecta	Dibromide	5×10^{6}	50% decrease in O_2 evolution*	....	Walsh 1972[379]
				Isochrysis galbana	Dibromide	1.5×10^{4}	50% decrease in growth	Measured as ABS. (525 mu) after 10 days	Walsh 1972[379]
				Phaeodactylum tricornutum	Dibromide	5×10^{6}	50% decrease in O_2 evolution*	....	Walsh 1972[379]
Paraquat [1,1′-dimethyl-4,4′-dipyridilium dichloride]	Not available	Unlikely	Unlikely	Dunaliella tertiolecta	Dichloride	[illegible]	50% decrease in O_2 evolution*	....	Walsh 1972[379]
				Isochrysis galbana	Dichloride	5×10^{3}	50% decrease in growth	Measured as ABS. (525 mu) after 10 days	Walsh 1972[379]
Trifluralin [α α α-Trifluoro-2,6-dinitro-N,N-dipropyl-p-toluidine]	Not available	Unlikely	Unlikely	Chlorococcum sp.	Technical acid	2.5×10^{3}	50% decrease in growth	Measured as ABS. (525 mu) after 10 days	Walsh 1972[379]
				Isochrysis galbana	Technical acid	2.5×10^{3}	50% decrease in growth	Measured as ABS. (525 mu) after 10 days	Walsh 1972[379]
				Phaeodactylum tricornutum	Technical acid	2.5×10^{3}	50% decrease in growth	Measured as ABS. (525 mu) after 10 days	Walsh 1972[379]
Cacodylic acid [Hydroxydi-methyl arsine oxide]	Not available	Unlikely	Unlikely	Insufficient data					
Insecticides									
Insecticides, total (includes rodenticides)	5.7×10^{8} lb								
Heptachlor [Heptachloro-tetrahydro-endo-methano-indene] (includes heptachlor epoxide)	Not available	Oysters (Bugg et al. 1967)[350]	Bald Eagles (Krantz et al. 1970)[365]	Thalassoma bifasciatum Bluehead	100%	0.8	LC-50	96 hr static lab bioassay	Eisler 1970b[357]
Endrin [Hexachloro-epoxy-octahydro-endo-endo-di-methanoraphthalene]	Not available	Oysters (Bugg et al. 1967,[350] Casper, 1967,[352] Rowe et al. 1971)[372]	Brown Pelican (Schreiber and Risebrough 1972,[374] Risebrough et al. 1968)[371]	Mugil cephalus Striped mullet	100%	0.3	LC-50	96 hr static lab bioassay	Eisler 1970b[357]
				Menidia menidia Atlantic silverside	100%	0.05	LC-50	96 hr static lab bioassay	Eisler 1970b[357]
Dieldrin [Hexachloro-epoxy-octahydro-endo-exo-dimethanonaphthalene]	Not available	Oysters (Bugg et al. 1967,[350] Casper, 1967,[352] Rowe et al. 1971)[372]	Bald eagles (Krantz et al. 1970)[365] Grey Whale, Sperm Whale (Wolman and Wilson 1970)[380] Brown Pelican (Schreiber and Risebrough 1972)[374]	Anguilla rostrata American eel	100%	0.9	LC-50	96 hr static lab bioassay	Eisler 1970b[357]
Aldrin [Hexachloro-hexa-hydro-endo-exo-dimeth-anonaphthalene]	Not available	Oysters (Bugg et al. 1967)[350]	Unlikely, converts to dieldrin (Korschgen 1970)[364]	Palaemon macrodactylus Korean shrimp	Technical	0.74 (0.51–1.08)	TL-50	96 hr static lab bioassay	Earnest (unpublished)[382]
Chlordane [Octochloro-hexahydro-methanoindene]	Not available	Oysters (Bugg et al. 1967)[350]	Expected	Palaemon macrodactylus Korean shrimp	100%	18 (10–30)	TL-50	96 hr static lab bioassay	Earnest (unpublished)[383]

* /O_2 evolution measured by Gilson differential respirometer on 4 ml of culture in log phase. Length of test 90 minutes.

TABLE IV-7—Presence and Toxicity of Organic Chemicals in the Marine System—Continued

Chemical (1)	U.S. production pounds, gal./yr (2)	Presence in sea water or marine organisms (3)	Trophic accumulation (4)	Most sensitive organisms tested (5)	Formulation (6)	Conc. (ppb active ingredient in water) (7)	Method of assessment (8)	Test procedure (9)	Reference (10)
Insecticides, cont.									
Strobane^{R} [polychlorinated terpenes]	Not available	Expected	Expected	Insufficient data for marine species					
Toxaphene [Chlorinated camphene]	Not available	Bay mussel (Modin, 1969);[367] Oysters (Bugg et al. 1967)[350]	Expected	Gasterosteus aculeatus threespine stickle-back	100%	7.8	TLM	96 hr static lab bioassay	Katz 1961[362]
DDT compounds	1.2×10^{8} lb.	Jensen et al. 1969,[360] Risebrough et al. 1968[371]							
p,p′-DDT [1,1,1-Trichloro-2,2-bis(p-chlorophenyl) ethane	· · ·	(References cited above)		Penaeus duorarum Pink shrimp	Technical 77%	0.12 0.17 (0.09–0.32)	TL-50 TL-50	28 day bioassay 96 hr intermittent flow lab bioassay	Nimmo et al. 1970[370] Earnest (unpublished)[382]
p,p′-DDD(p,p′-TDE) [1,1-Dichloro-2,2-bis (p-chlorophenyl)ethane	· · ·			Palaemon macrodactylus	99%	2.5 (1.6–4.0)	TL-50	96 hr intermittent flow lab bioassay	Earnest (unpublished)[382]
p,p′-DDE [1,1-Dichloro-2,2-bis(p-chlorophenyl) ethylene		(References cited above)		Falco peregrinus Peregrine Falcon			Eggshell thinning	DDE in eggs highly correlated with shell thinning	Cade et al. 1970[351]
Mirex [Dodecachloro-octahydro-1,3,4-metheno-2H-cyclobuta[cd]pentalene]	Not available	Expected	Expected	Penaeus duorarum Pink shrimp	Technical	1.0	100% paralysis/death in 11 days	Flowing water bioassay	Lowe et al. 1971[365]
Benzene hexachloride [Hexachlorocyclohexane]	Not available	Southern hemisphere sea birds (Tatton and Ruzicka 1967)[375]		Penaeus setiferus White shrimp	8.1%	2.8	TLM	24 hr static lab bioassay	Chin and Allen 1958[353]
Lindane [gamma-hexachlorocyclohexane]	Not available	Oysters (Bugg et al. 1967,[350] Casper 1967)[352]	Expected Sand shrimp	Crangon septemspinosa Sand shrimp	100%	5	LC-50	96 hr static lab bioassay	Eisler 1969[355]
				Pagurus longicarpus Hermit crab	100%	5	LC-50	96 hr static lab bioassay	Eisler 1969[355]
Endosulfan [Hexachlorohexahydro-methano-benzo-dioxathiepin-3-oxide] (Thiodan^{R})	Not available	Bay mussel (Koeman and Genderen 1970)[363]	Sandwich Tern, Common Eider (Koeman and Genderen 1970)[363]	Palaemon macrodactylus Korean shrimp	96%	3.4 (1.8–6.5)	TL-50	96 hr intermittent flow lab bioassay	Earnest (unpublished)[382]
Methoxychlor [1,1,1-Trichloro-2,2-bis(p-methoxy-phenyl)ethane]	Not available	Oysters (Bugg et al. 1967)[350]	Unlikely	Palaemon macrodactylus	89.5%	0.44 (0.21–0.93)	TL-50	96 hr static lab bioassay	Earnest (unpublished)[382]
Carbaryl (Sevin) [1-naphthyl-N-methylcarbamate]	Not available	Unlikely	Unlikely	Palaemon macrodactylus	100%	7.0 (1.5–28)	TL-50	96 hr intermittent bioassay	Earnest (unpublished)[382]
				Cancer magister Dungeness crab	80%	6	Prevention of hatching and molting	24 hr static lab bioassay	Buchanan et al. 1970[349]
Coumaphos (Co-ral) [O,O-Diethyl-O-(3-chloro-4-methyl-2-oxo-2H-1-benzopyran-7-yl)-phosphorothioate]	Not available	Unlikely	Unlikely	Crassostrea virginica American oyster		110	TLM	48 hr static lab bioassay	Davis and Hidu 1969[354]
Diazinon [O,O-Diethyl-O-(2-isopropyl-4-methyl-6-pyrimidinyl)phosphorothioate]	Not available	Unlikely	Unlikely	Insufficient data					
Parathion [O,O-Diethyl-O-p-nitrophenyl-phosphorothioate]	Not available	Unlikely	Unlikely	Cyprinodon variegatus Sheepshead minnow		10	Acetylcholinesterase activity in control vs. expt groups. Control = 1.36; Expt. = 0.120	72 hr static exposure	Coppage (unpublished)[381]
Dursban [O,O Diethyl-O-3,5,6-trichloro-2-pyridyl-phosphorothioate]	Not available	Unlikely	Unlikely	Palaemon macrodactylus Korean shrimp		0.01 (0.002–0.046)	TL-50	96 hr intermittent flow bioassay	Earnest (unpublished)[382]
Fenthion [O,O-Dimethyl-O-(4-methylthio-m-tolyl) phosphorothioate] (Baytex)	Not available	Unlikely	Unlikely	Palaemon macrodactylus		3.0 (1.5–60)	TL-50	96 hr intermittent flow bioassay	Earnest (unpublished)[382]
Methyl parathion [O,O,-Dimethyl-O-p-nitrophenyl-phosphorothioate]	5.1×10^{7} lb	Unlikely	Unlikely	Crangon septemspinosa Sand shrimp	100%	2	LC-50	96 hr static lab bioassay	Eisler 1969[355]
Guthion [O,O-Dimethyl-S-(4-oxo-1,2,3-benzotriazino-3-methyl)phosphorodithioate]	Not available	Unlikely	Unlikely	Gasterosteus aculeatus threespine stickle-back	93%	4.8	TLM	96 hr static lab bioassay	Katz 1961[362]
Dioxathion (Delnav) [2,3-p-Dioxane-S,S-bis(O,O-diethylphosphorodithioate)]	Not available	Unlikely	Unlikely	Crangon septemspinosa Sand shrimp	100%	38	LC-50	96 hr static lab bioassay	Eisler 1969[355]
				Fundulus heteroclitus Mummichog	100%	6	LC-50	96 hr static lab bioassay	Eisler 1970a[356]
				Menidia menidia Atlantic silverside	100%	6	LC-50	96 hr static lab bioassay	Eisler 1970b[357]

***TABLE IV-7—Presence and Toxicity of Organic Chemicals in the Marine System*—Continued**

Chemical (1)	U.S. production pounds, gal./yr (2)	Presence in sea water or marine organisms (3)	Trophic accumulation (4)	Most sensitive organisms tested (5)	Formulation (6)	Conc. (ppb active ingredient in water) (7)	Method of assessment (8)	Test procedure (9)	Reference (10)
Insecticides, cont.									
Phosdrin [1-methoxycarbonyl-1-propen-2-yl dimethylphosphate]	Not available	Unlikely	Unlikely	Crangon sepemspinosa Sand shrimp	100%	11	LC-50	96 hr static lab bioassay	Eisler 1969[355]
Malathion [S-(1,2-dicarbethoxyethyl)-O,O-dimethyldithiophosphate]	Not available	Unlikely	Unlikely	Thalasomma bifasciatum Bluehead	100%	27	LC-50	96 hr static lab bioassay	Eisler 1970b[357]
Phosphamidon [2-Chloro-N,N-diethyl-3-hydroxy-crotonamide dimethyl phosphate]	Not available	Unlikely	Unlikely	Insufficient data					
Phorate [O,O Diethyl-S-([Ethylthio]methyl)-phosphorodithioate]	Not available	Unlikely	Unlikely	Cyprinodon variegatus Sheepshead minnow		5	Acetylcholinesterase activity** in control vs expt. groups. Control = 1.36; Expt. = 0.086	72 hr static exposure	Coppage (unpublished)[381]
DDVP [O,O Dimethyl-O-(2,2-dichlorovinyl)phosphate]	Not available	Unlikely	Unlikely	Crangon septemspinosa Sand shrimp		4	LC-50	96 hr static lab bioassay	Eisler 1969[355]
Trichlorfon [O,O-Dimethyl-1-hydroxy-2,2,2-trichloroethylphosphonate] (Dipterex)	Not available	Unlikely	Unlikely	Crassostrea virginica American oyster		1,000	TLM	48 hr static lab bioassay	Davis and Hidu 1969[354]
TEPP [Tetraethyl pyrophosphate]	Not available	Unlikely	Unlikely	Crassostrea virginica		$>1\times10^4$	TLM	14 day static lab bioassay	Davis and Hidu 1969[354]
Related products									
DBCP [1,2-Dibromo-3-chloropropane] (Nemagon[12])	8.6×10^6 lb	Unknown	Unknown	Mercenaria mercenaria Hard clam		780	TLM	12 day static lab bioassay	Davis and Hidu 1969[354]
Methyl bromide	2.0×10^7 lb	Unknown	Unknown	Insufficient data					
TAR AND TAR CRUDES									
Benzene	1.2×10^9 gal.	Unknown	Unknown	Insufficient data					
Toluene	7.6×10^8 gal.	Unknown	Unknown	Insufficient data					
Xylene	3.8×10^8 gal.	Unknown	Unknown	Insufficient data					
Naphthalene	8.5×10^8 gal.	Unknown	Unknown	Insufficient data					
PLASTICIZERS									
Phthalic anhydride esters, total	8.8×10^8 lb.	Expected	Unknown	Insufficient data					
Adipid acid esters, total	6.6×10^7	Unknown	Unknown	Insufficient data					
SURFACE-ACTIVE AGENTS									
Dodecylbenzenesulfonates, total	5.7×10^8 lb. (1968)	Unknown	Unlikely	Insufficient data					
Ligninsulfonates, total	4.4×10^8 lb.	Unknown	Unknown	Insufficient data					
Nitrilotriacetic acid	Not available	Unknown	Unlikely	Cyclotella nana	Monohydrated sodium salt	5×10^3	38% growth as compared to controls	72 hr static lab bioassay	Erickson et al. 1970[360]
				Homarus americanus American lobster	Monohydrated sodium salt	1×10^5	100% mortality	7 day static lab bioassay	NMWQL 1970[378]
HALOGENATED HYDROCARBONS									
Carbon tetrachloride	7.6×10^8 lb (1968)	Unknown	Unlikely	Insufficient data					
Dichlorodifluoromethane	3.3×10^8 (1966)	Unknown	Unlikely	Insufficient data					
Ethylene dichloride	4.8×10^9 (1968)	Expected	Unlikely	Insufficient data					
Aliphatic chlorinated hydrocarbon wastes of vinyl chloride production	3×10^7 lb (estimated as 1% of vinyl chloride production)	Surface waters and marine organisms of North Atlantic and North Sea (Jensen et al. 1970)[361]	Unknown	Gadus morrhua Cod	67% 1,1,2-trichloroethane, 20% 1,2-diethane	10,000	LC-50	10 hr lab bioassay	Jensen et al. 1970[361]
Polychlorinated biphenyl	Not available	Jensen et al. 1969[369], Risebrough et al. 1968[371]		Penaeus duorarum Pink shrimp	Aroclor 1254	0.94	51% mortality	15 day chronic exposure in flowing seawater	Nimmo et al. 1971[370]
Polychlorinated terphenyl	Not available	Expected	Expected	Insufficient data					
Brominated biphenyls	Not available	Unknown	Expected	Insufficient data					
CYCLIC INTERMEDIATES									
Monochlorobenzene	6.0×10^8 lb	Expected	Unlikely	Insufficient data					
Phenol	1.7×10^9 lb	Expected	Unlikely	Mercenaria mercenaria Hard Clam	....	5.3×10^4	TLM	48 hr static lab bioassay	Davis and Hidu 1969[354]
MISCELLANEOUS CHEMICALS									
Tetraethyl lead	4.9×10^8	Unlikely	Unlikely	Insufficient data					

** ACh hydrolysed/hr/mg brain.

whether or not the compound has been detected in sea water or in marine organisms. Compounds which have been detected are of greater immediate concern than those which have not. Frequently, because of their low solubility in water, some of the non-polar compounds which are biologically accumulated can be detected in an organism but not in the water itself. (c) The fourth column, trophic accumulation, indicates whether the compound has been shown to pass through the food web from prey species to predator. Compounds that are so accumulated are of greater concern than compounds of comparable toxicity which are not. Finally, the species thought to be most sensitive to the compound are indicated in the final columns with reference to original studies in the scientific literature. These data are useful as a guide only and are not sufficient in themselves for definitive evaluation of the environmental significance of each compound.

The report, "The Effects of Chemicals on Aquatic Life, vol. 3, Environmental Protection Agency, Water Quality Office, 1971," has been useful as a guide to the available toxicity data of industrial chemicals on marine organisms. Appendix III–Table 6 is a compendium of data on toxicity of pesticides to marine organisms. These sources are incomplete and should be continually revised.

Bases for Recommendations

1. In order to provide an adequate level of protection for commercially important marine species and for species considered important in the maintenance of stability of the ecosystem, an application factor of one one-hundredth (0.01) is used when pesticides or organic wastes that are not trophically accumulated in food webs are applied or released in estuarine or marine environments. This factor is arbitrary and was derived from data available on marine and freshwater organisms. (See Section III, p. 121.) It assumes that a concentration of one one-hundredth (0.01) of that causing harm to the most sensitive species to be protected will not damage this species or the ecosystem. Future studies may show that the application factor must be decreased or increased in magnitude.

2. The application factor may also be used for the compounds that are trophically accumulated in food webs in order to protect fish and invertebrates to which these compounds are toxic. It cannot be used, however, to protect fish-eating birds and mammals which may trophically accumulate these compounds from their prey species, in part because sublethal effects such as eggshell thinning and hormone imbalance may adversely affect reproductive capacity and therefore the long term survival of populations. Levels that would protect fish-eating birds and mammals against the effects of compounds that are trophically accumulated from prey species are given in the discussion of Marine Wildlife (*see* pp. 224–228).

The recommendations below apply to all organics of both proved and potential toxicity.

Recommendations

In general, marine life with the exception of fish-eating birds and mammals should be protected where the maximum concentration of the chemical in the water does not exceed one one-hundredth (0.01) of the LC50 values listed in Column 7, Table IV–7, pp. 265–268. If new data indicate that an ecosystem can adequately degrade a particular pollutant, a higher application factor for this pollutant may be used.

In order to maintain the integrity of the ecosystem to the fullest possible extent, it is essential to consider effects on all non-target organisms when applying pesticides to estuarine habitats in order to control one or more of the noxious species. For those occasions when chemicals must be used, the following guidelines are offered:

- **a compound which is the most specific for the intended purpose should be preferred over a compound that has broad spectrum effects;**
- **a compound of low persistence should be used in preference to a compound of greater persistence;**
- **a compound of lower toxicity to non-target organisms should be used in preference to one of higher toxicity;**
- **water samples to be analyzed should include all suspended particulate and solid material: residues associated with these should therefore be considered as present in the water;**
- **when a derivative such as p,p′-DDE or 1-napthol is measured with or instead of the parent compound, the toxicity of the derivative should be considered separately: if the toxicity of a derivative such as an ionic species of a pesticide is considered equivalent to that of the original parent compound, concentrations should be expressed as equivalents of the parent compound.**

It is recommended that the chemicals listed in Table IV–7 and all chemicals subsequently added to this list be considered as toxic organic compounds potentially harmful to the marine environment. It is emphasized that the data in Table IV–7 are not sufficient in themselves for final evaluation of the environmental significance of each compound.

OXYGEN

An extensive review and discussion of the present information on biological responses to variations in dissolved oxygen has been published recently by Doudoroff and

Shumway (1970).[383] This review has been used in developing oxygen recommendations by both the Freshwater and Marine Panels in their reports. On the basis of this large body of information, recommendations for "levels of protection" for freshwater fish populations have been developed. Estuarine and marine organisms have not been studied as extensively, and the present information is inadequate for satisfactory analysis of the response of communities to temporal and spatial variations in dissolved oxygen concentrations.

The generalizations presented by the Freshwater Panel appear to be valid, with qualifications, for estuarine and marine situations.

1 A reduction in dissolved oxygen concentration reduces the rate of oxygen uptake by aquatic plants and animals. However, as noted by Doudoroff and Shumway, the observed response of many organisms under laboratory conditions measured in such terms as growth rate, swimming speed, or hatching weight, shows fractional or percentage reductions that approximately correlate with the logarithm of the deviation of the dissolved oxygen concentration from equilibrium with the atmosphere, under conditions of *constant* dissolved oxygen concentrations. Thus, reduction in the dissolved oxygen concentration by 1 mg/l from the saturation value has much less effect than reduction by 1 mg/l from the 50 per cent of saturation value.

2 The non-threshold character of these responses means that some risk of effect on the aquatic populations is associated with any reduction in the dissolved oxygen concentrations. As noted above, the risk of damage increases as dissolved oxygen concentrations decrease from saturation values. Selection of risk acceptance is a social and economic evaluation involving other uses of any particular environment that must precede recommendations derived using the risk acceptance and the pertinent scientific information.

3 Consideration of the effects of dissolved oxygen concentrations on aquatic life must include the responses of developing eggs and larvae, as well as the maturing and adult individuals. Species that have limited spawning areas should be identified and the biological risk of decreased oxygen concentrations evaluated accordingly.

For estuaries and coastal waters, consideration must be given to the distribution of dissolved oxygen with depth, since even under natural conditions low oxygen concentrations may be found in the deeper waters. Special consideration should be given to estuary type, topography, currents, and seasonal development of pycnoclines.

Many estuaries and coastal regions are highly productive, and the characteristic pattern with photosynthesis in the upper-water layer or adjacent marshes leads to large population densities in the upper layers and loss of oxygen to the atmosphere from the supersaturated surface waters or the marsh plants. Subsequent decomposition of these organisms and their wastes in the deeper waters leads to oxygen depletion. Several deeper coastal plain estuaries and fjords show oxygen depletion from this sequence. Addition of mineral and organic plant nutrients to such regions may intensify the production and subsequent decomposition processes. The effects of particular additions will depend on the water depths and rate of vertical mixing, and it is necessary to construct an oxygen balance model for each case. Sewage treatment that consists of partial or nearly complete mineralization of the organic materials may still produce a discharge that will damage the aquatic system, i.e., an amount of organic matter nearly equal in oxygen demand to the original sewage is produced in the environment. The principal effect of many "secondary" treatment systems is the trading of an intense local effect near the outfall for a more widespread effect at greater distances. One of the major considerations in defining water quality recommendations for nutrients in any estuarine or coastal region should be the risk associated with oxygen depletions from increased production. Deliberate moderate additions of nutrients to increase the yield of some fishery should also give due regard to this secondary effect.

Recommendation

Each proposed change in the dissolved oxygen concentration in estuaries and coastal waters should be reviewed for risk of damage to aquatic life. The limited laboratory data and field observations on marine organisms suggest that easily observed effects, which are in many cases deleterious, occur with dissolved oxygen concentrations of 4 to 5 mg/l as daily minimum values for periods of several days. As a guideline, therefore, reduction of the dissolved oxygen concentration to values below 4 mg/l can be expected to change the kinds and abundances of the aquatic organisms in the affected volume of water and area of bottom. Particular attention should be directed toward identifying species with restricted spawning and nursery areas and conservatism should be used in applying guidelines to these areas. (See the expanded discussion in Section III, pp. 131–135.)

RADIOACTIVE MATERIALS IN THE AQUATIC ENVIRONMENT

This section considers radioactivity in all surface waters inhabited by plants and animals including fresh, estuarine, and marine waters of the U.S. The subject matter pertains primarily to the impact of environmental radioactivity on aquatic organisms, although it also contains some discussion of human radiation exposure from aquatic food chains. A recent report by the National Academy of Sciences (1971)[397] presented a review of radioactivity in the marine environment, and that review has been used extensively in the preparation of this report.

Characteristics and Sources of Radioactivity

Radiation is the energy emitted spontaneously in the process of decay of unstable atoms of radioisotopes. This energy can exist either in the form of electromagnetic rays or subatomic particles and cannot be detected by man's senses. Radiation can be detected, however, by means of electronic instruments, and quantities present at very low levels in the environment can be measured with remarkable accuracy. Radioactivity which occurs naturally in the environment originates from primordial radioisotopes and their decay products (daughters) and from reactions between cosmic rays from outer space and elements in the atmosphere or in the earth. Some of the more abundant primordial radioisotopes in terms of their radioactivity are potassium (^{40}K), palladium (^{234}Pd), rubidium (^{87}Rb), uranium (^{238}U) and thorium (^{237}T), the first accounting for 90 per cent of the natural radiation in the oceans. While beryllium (^{7}Be) is the most abundant radioisotope produced by cosmic rays, carbon (^{14}C) and hydrogen (^{3}H) (tritium) are biologically the most interesting. The presence of natural radioactivity was unknown until 1896 when Becquerel discovered uranium. Until the development of the atomic bomb during World War II, virtually all of the radioactivity on earth came from natural sources.

The first man-made radioisotopes were not released into the environment in any significant amounts until the atomic bomb was tested and used in war even though the uranium 235 atom was first split (fissioned) by neutron bombardment in 1938. While the release of radioisotopes was drastically reduced with the halting of nuclear weapons testing in the atmosphere by signatories of the test ban treaty, radioactive wastes continue to be released from nuclear powered ships and submarines, nuclear power plants, nuclear fuel reprocessing plants, and to a lesser extent from laboratories and hospitals. Two methods have been used in handling radioactive wastes. High levels have been concentrated and held in special storage tanks, while low levels of radioactive wastes in small volumes have been diluted and dispersed in the aquatic environment particularly in the oceans. Some manmade radioisotopes, such as strontium 90 and cesium 137, are the debris of split atoms and are called fission products. Other radioisotopes, such as zinc 65 and cobalt 60, are activation products, produced when stray neutrons from the fission process strike the atoms of stable elements.

Cycling of Radioactive Materials The physical, chemical, and physiological behavior of radioisotopes is essentially identical with that of the stable isotopes of the same element—at least until disintegration occurs. It should be pointed out, however, that in some instances the physical and chemical states of a radioisotope introduced into the aquatic environment may vary from that of the stable element in water. At the time of disintegration, the decaying atoms change into different types of atoms of the same element or into atoms of a different element. If the behavior of a particular element in an ecosystem is known, the behavior of the radioisotopes of that element can be predicted. The reverse also is true, and radioisotopes can serve as excellent tracers in following the movement of elements through complex environmental systems. Radioactive wastes in the aquatic environment may be cycled through water, sediment, and the biota. Each radioisotope tends to take a characteristic route and has its own rate of movement through various temporary reservoirs. The route taken by tritium is different from that of other radioisotopes. Tritium becomes incorporated in the water molecule and cannot be removed by present waste treatment practices. It is not concentrated appreciably by either biota or sediments.

When radioactive materials enter surface waters they are diluted and dispersed by the same forces that mix and distribute other soluble or suspended materials (National Academy of Sciences 1957).[393] The dominant forces are mechanical dilution that mixes radioisotopes in the waste stream as it leaves an outfall structure; advection and turbulent diffusion that mix materials in the receiving waters; and major transport currents that move masses of water over relatively long distances. On the other hand, precipitation and sedimentation tend to restrict the area of dispersion. When first introduced into fresh or marine water, a substantial part of the materials present in radioactive wastes becomes associated with solids that settle to the bottom, and many of the radioisotopes are bound chemically to the sediments. The sediments may also be moved geographically by currents. Even though in some instances sediments remove large quantities of radioisotopes from the water, and thus prevent their immediate uptake by the biota, this sediment-associated radioactivity may later leach back to the water and again become available for uptake by the biota.

Plants and animals, to be of any significance in the passage of radioisotopes through a food web in the aquatic environment, must accumulate the radioisotope, retain it, be eaten by another organism, and be digested. Radioisotopes may be passed through several trophic levels of a food web, and concentrations can either increase or decrease from one trophic level to the next, depending upon the radioisotope and the particular prey-predator organisms. This variation among trophic levels occurs because different organisms within the same trophic level have different levels of concentration and different retention times, which depend upon their metabolism or capacity to concentrate a given radioisotope. The concentration of a radioisotope by an organism is usually discussed in terms of a concentration factor: the ratio of the concentration of the radioisotope in the organism to that in its source, that is, the amount in water or food. Radioisotopes with short half-lives are less likely to be highly concentrated in the higher trophic levels of the food chain because of the time required to move from the water to plants, to herbivores, and eventually to carnivores. Organisms that concentrate radioisotopes to a high level and retain them for long periods of time have been referred to as "biological indicators for radioactivity." These organisms are of value in showing the presence of radioactive materials even though the concentrations in the water may be less than detectable limits.

Exposure Pathways

The radiation emitted by radioisotopes that are present in aquatic ecosystems can irradiate the organisms in many different ways. In order to evaluate the total radiation dose received by the aquatic organisms, and thus the risk of their being injured, all sources of exposure must be considered. These sources include both natural and man-made radiation, both external and internal.

Major Sources of External Radiation 1 Radioisotopes in the surrounding water that tend to remain in solution, or at least suspended in the water, become associated more readily with aquatic organisms than the radioisotopes that settle out.

2 Radioisotopes present on or fixed to sediments are significant to aquatic life, particularly to benthic organisms in the vicinity of existing major atomic energy plants.

3 Radioisotopes attached to the outer surfaces of organisms are of greater significance to micro-organisms, which have a larger surface-to-volume ratios, than shellfish or fish.

4 Cosmic-rays are of relatively minor importance to aquatic life that lives a few feet or more below the water surface, because of the shielding afforded by the water.

Major Sources of Internal Radiation 1 Radioisotopes in the gastrointestinal tract frequently are not assimilated, but during their residence in the tract expose nearby internal organs to radiation.

2 Assimilated radioisotopes are absorbed from water through the integument or from food and water through the walls of the gastrointestinal tract, metabolized, and are incorporated into tissues where they remain for varying periods of time. Aquatic plants, including algae absorb radioactive materials from the ambient water and from the interstitial water within the sediments.

It is difficult to measure the amount of radiation absorbed by aquatic organisms in the environment because they are simultaneously irradiated by radioisotopes within their body, on the surface of their body, in other organisms, in the water, and in sediments. Exposure thus depends on an organism's position in relation to the sediments and to other organisms, and to movement of some species in and out of the contaminated area.

Biological Effects of Ionizing Radiation

Ionizing radiation absorbed by plant and animal tissue may cause damage at the cellular and molecular levels. The degree of radiation damage to an organism depends upon the source (external or internal), the type (electromagnetic or particulate), the dose rate (intensity per unit of time), and the total dose. Possible effects to the individual organism may include death, inhibition or stimulation of growth, physiological damage, changes in behavioral patterns, developmental abnormalities, and shortening of life span. In addition, the extent of biological damage from radiation can be modified by environmental stresses such as changes in temperature and salinity. Under certain conditions, irradiation can cause gross pathological changes which are easily observed, or more subtle changes which are difficult or impossible to detect. In addition to somatic changes which affect the individual, genetic changes also may occur which may affect the offspring for many generations. At one time, it was widely believed that there was a threshold radiation dose below which damage did not occur, but now the consensus of most radiobiologists is that any increase over background radiation will have some biological effect. While the non-existence of a threshold dose is difficult to prove, most radiation biologists agree that even background levels of radiation from primordial radioisotopes and cosmic rays have resulted in some genetic changes over the ages. These radiation-induced changes usually constitute less than 1 per cent of all spontaneously occurring mutations (Asimov and Dobzhansky 1966).[384]

The amount of radiation absorbed by an organism can be expressed in various ways. The rad (radiation absorbed dose) is the unit used to measure the absorbed dose of radiation and refers to the absorption of 100 ergs of energy per gram of irradiated material. Because a rad of alpha or neutron radiation produces greater biological damage than a rad of gamma radiation, another unit called the rem (roentgen equivalent man) also is used. To obtain the rem, or dose equivalent, the number of rads absorbed by the tissue is multiplied by the quality factor and other necessary modifying factors to compensate for the effects of different types of radiation. The acute doses of radiation required to produce somatic damage to many species of aquatic organisms have been established within broad limits (National Academy of Sciences 1971).[397] Some bacteria and algae can tolerate doses of many thousands of rads, but the mean lethal dose (LD^{50} 30 days) for fish is in the range of several hundred to a few thousand rads. Eggs and early developmental stages are more sensitive than are adults. By comparison, the mean lethal dose for humans is about 300 rads.

The acute mean lethal dose has little value in placing restrictions on the amounts of radioactive material present in aquatic environments. Much more meaningful is the highest level of chronic exposure that results in no demonstrable damage to aquatic populations. A vast amount of research on dose-effect relationships for warm-blooded animals has led to the recommendations on human radiation exposure. People who work with radiation may receive no more than 5 rem in any one year. The recommended limit for the general public is 0.5 rem in one year for individuals but is restricted to only 0.17 rem per year as an average for populations. The lower level permitted for populations is to reduce the possibility of genetic changes becoming established.

Compared with the experimental data available for warm-blooded animals, only a meager amount of information is available on chronic dose-effect relationships for aquatic forms. The preponderance of available data indicates, however, that no effects are discernible on either individual aquatic organisms or on populations of organisms at dose rates as high as several rads per week. In populations of wild species, genetic damage may be removed by natural selection and somatically weakened individuals are probably eaten by predators. Consequently, aquatic organisms adversely affected by radiation are not readily recognized in the field.

The natural populations of fish that have probably sustained the greatest exposure to man-made radioactive materials are those near major atomic energy installations, for example, in the Columbia River near Hanford; in White Oak Creek and White Oak Lake, near Oak Ridge; and in the Irish Sea near Windscale, England. Small fish which received chronic irradiation of about 10.9 rads per day from radioisotopes in the sediments of White Oak Creek produced larger broods but with a higher incidence of abnormal embryos (Blaylock and Mitchell 1969).[385] Chironomid larvae living in the bottom sediments and receiving about five rads per week had an increased frequency of chromosomal aberrations but the abundance of the worms was not affected. The stocks of plaice in the vicinity of the Windscale outfall have been unaffected by annual dose rates of about 10 rads per year—primarily from the bottom sediments (Ministry of Agriculture, Fisheries and Food 1967).[392] Columbia River salmon spawning in the vicinity of the Hanford outfalls have been unaffected by doses in the range of 100 to 200 millirads per week (Watson and Templeton *in press*).[402] These observations on chronic exposure of aquatic organisms provide a subjective assessment of radiation sensitivities in natural populations but are not sufficiently definitive to form the basis for the development of water quality recommendations.

Restrictions on Radioactive Materials

The amounts of radioactive materials present in water must be restricted in order to assure that populations of organisms are not damaged by ionizing radiation and also to limit the amount of radioactive material reaching man via aquatic food chains. Permissible rates of intake of the various radioisotopes by man have been calculated so that the resulting annual dose is no greater than the recommended limit. Therefore, when the rate of consumption of aquatic organisms is determined, e.g., pounds of fish or shellfish per year, maximum concentrations of radionuclides permissible in the edible parts of the organisms can be computed. These maximum concentrations are well below the concentrations which have produced detectable effects on natural aquatic populations. It is probable that the aquatic environment will be protected by the restrictions currently imposed on the basis of human health.

The regulations which serve to protect man from radiation exposure are the result of years of intensive studies on the biological effects of radiation. Vast amounts of information have been considered by the International Commission on Radiological Protection (ICRP) (1960,[389] 1964,[390] 1965[391]), the National Council on Radiation Protection and Measurements (NCRP) (1959,[398] 1971[399]), and the U.S. Federal Radiation Council (FRC) (1960,[387] 1961[388]), in developing recommendations on the maximum doses of radiation that people may be allowed to receive under various circumstances or that may occur in water. The Drinking Water Standards (U.S. Department of Health, Education and Welfare, Public Health Service 1962[400]) and the Code of Federal Regulations (1967)[386] are responsive to the recommendations of the FRC, ICRP, and NCRP, and provide appropriate protection against unacceptable radiation dose levels to people where drinking water is the only significant source of exposure above natural background. Where fish or other fresh or marine products that have accumulated radioactive materials are used as food by humans, the concentrations of the radiosiotopes in the water must be further restricted to ensure that the total intake of radioisotopes from all sources will not exceed the recommended levels.

Conclusions

Previous attempts to restrict radioactive discharges to marine environments have resulted in recommended maximum permissible concentrations in sea water (National Academy of Sciences 1959a,[394] 1959b,[395] 1962,[396] 1971[397]). These recommendations are most useful as a first approximation in predicting safe rates of discharge of radioactive wastes, but their applicability as water quality recommendations is limited and they are not intended for general use in fresh or estuarine waters where the concentrations of a great variety of chemical elements vary widely.

Three approaches to the control of levels of radioactivity in the aquatic environment have been used: (1) controlling the release of radioactivity based upon the specific activity approach—the ratio of the amount of radioactive isotope present to the total amount of the element (microcuries per milligram) (National Academy of Sciences 1962),[396] (2) relating the effects of radiation upon aquatic organisms caused by a given concentration of a radioisotope or combinations of radioisotopes in the water, and (3) restricting concentrations of radioisotopes to those permitted in water and food for human consumption.

Since concentrations of stable elements vary from one body of water to another, and with time, and since adequate data are not available to relate effects of radiation upon aquatic organisms to specific levels of radioactivity in the water, restrictions contained in the Code of Federal Regulations (1967)[386] on liquid effluents are considered adequate to safeguard aquatic organisms.

Because it is not practical to generalize on the extent to which many of the important radioisotopes will be concentrated by aquatic organisms, nor on the extent to which they will be used for food by people, no attempt is made here to specify maximum permissible concentrations (MPC) for water in reference to uptake by the organisms. Rather, each case requires a separate evaluation that takes into account the peculiar features of the region. Such an evaluation should be approved by an agency of the State or Federal Government in each instance of radioactivity contamination in the environment. In each particular instance of proposed contamination, there must be a determination of the organisms present, the extent to which these organisms concentrate the ratioisotopes, and the extent to which man uses the organisms as food. The rates of release of radioisotopes must be based on this information.

Recommendation

Aquatic organisms concentrate radioisotopes to various degrees in their tissues. The concentration in sea water should be low enough so that the concentration in any aquatic species will not exceed Radiation Protection Guides of the U.S. Federal Radiation Council (1961)[401] for organisms harvested for use as human food. This recommendation is based upon the assumption that radiation levels which are acceptable as human food will not injure the aquatic organisms including wildlife.

SEWAGE AND NUTRIENTS

Magnitude of the Problem

The discharge of municipal sewage is a major factor affecting the water quality of receiving systems. Because the amount of municipal waste produced is directly related to the human population, the unit emission rates together with information on the number of people using a system provide an accurate estimate of the load that is imposed on a particular estuary or section of coastal water.

The effect of sewage discharges on water quality varies widely and depends on (1) its composition and content of toxic materials, (2) the type and degree of treatment prior to discharge, (3) the amount released, (4) the hydrodynamics of the receiving waters, and (5) the response of the ecosystem. Increasing human population and affluence have resulted in increasing amounts of domestic and industrial wastes. However, because the kind and degree of treatment often can be improved, it should be possible to cope with this pollution problem and to maintain or improve the quality of the marine environment.

In most cases the discharge of sewage effluent is intentional and the source of sewage and sewage treatment products entering marine ecosystems can be described more accurately than the sources of other pollutants entering the ecosystem. The volume of discharges and certain aspects of their composition, specifically, the amount of organic matter and the inorganic nutrients, can be monitored continuously by existing automated methods. Average values for some important constituents and their emission rates in a densely populated coastal area are given in Table IV-8.

TABLE IV-8—Average Sewage Emissions for a Densely Populated Area

Constituent	Mass emission rate (tons/day)[a]	Unit emission rate (lb/capita/day)
Dissolved solids	3,600	1.03
Suspended solids	565	0.162
BOD	560	0.160
Total nitrogen (N)	165	0.047
Phosphate (Po$_4$)	100	0.029

[a] For 700 mgd of sewage; population of 7 million.
NAS-NRC Committee on Oceanography 1970[417].

Runoff from agriculture areas is an important factor in the nutrient enrichment of freshwater systems, but it is less important to marine systems because relatively fewer farms are concentrated on estuaries and coasts. Nevertheless, agricultural practices should be considered. Pesticides, fertilizers and animal wastes may be carried by rivers into estuaries. Runoff from duck farms was involved in a study on excessive nutrient enrichment by Ryther (1954).[422] Commoner (1970)[404] has emphasized that in the United States during the last twenty-five years the amount of nitrogen used in agriculture has increased fourteenfold while the amount of nitrogen released via sewage has increased only seventy per cent.

In addition to degradable organic materials derived from fecal and food wastes, municipal sewage also contains a wide variety of "exotic" or synthetic materials that are nondegradable or degrade slowly and only under special conditions (e.g., petroleum residues, dissolved metals, detergents, dyes, solvents, and plasticizers). Some of these adversely affect the biota of receiving waters, and many interfere with the biological degradation of organic matter either in the treatment plant or in the environment. Because waste treatment technology currently in use is designed to treat the fecal and food materials derived from organic wastes, an operational definition of municipal sewage "exotics" is all those materials not derived from fecal or food sources. If the exotic materials accumulate in the receiving ecosystem, the capacity for recycling of the degradable organic materials may be reduced.

Oxygen Depletion

Efficient biological degradation of organic materials requires dissolved oxygen, and overload of sewage in receiving waters can result in oxygen depletion and secondary effects such as objectionable odors, plant and animal die-off, and generally decreased rates of biological degradation. Such effects can also be created by excessive algal growth and subsequent die-off.

The most widely used method for estimating the organic pollution load of a waste is the 5 day Biochemical Oxygen Demand Test (BOD_5). Discussions of the test (Fair et al. 1968,[407] Standard Methods 1971[403]) and its limitations (Wilhm and Dorris 1968[426]) are available. Among the important limitations of the BOD_5 are: it does not indicate the presence of organics which are not degraded under the prescribed conditions; it assumes that no toxic or inhibitory materials will affect microbial activity; and it does not measure the nitrogeneous oxygen demand of the organic waste. The chemical oxygen demand (COD) is an alternate procedure for determining the amount of oxidizable material in a water sample. However, it does not indicate the nature of biological oxygen consumption in a given time, and it does not distinguish between inorganically and organically oxidizable materials. Both BOD_5 and COD measurements must be recognized as being only partial descriptions of the sewage load of a receiving water. While BOD_5 and COD measurements are useful for evaluating treatment systems, these two measurements do not adequately assess the environmental impact of a given sewage load (Wilhm and Dorris 1968).[426]

Excessive Nutrient Enrichment

Marine plants, like those on land and in fresh water, require fertilizing elements essential for their growth and reproduction. These essential elements are natural constituents of municipal sewage and the amount that can be added to the marine environment without deleterious effect is determined by the stimulated growth of aquatic plants. Even if the major share of the organic material is removed from the sewage in treatment plants, the growth of normal marine plants can increase if the fertilizing elements present in sewage are added to the environment. Sewage treatment plants are designed to remove the organic material and the suspended solids and to decrease the bacterial population by disinfection. In most cases, this is done by processes that release or "mineralize" the plant nutrients which then stimulate the growth of algae in the receiving waters. In only a few cases have efforts been made to remove these fertilizers from the effluent to prevent or reduce the excessive growth of plants in the aquatic environment.

In the marine environment, growth of phytoplankton is commonly limited by the availability of essential nutrients, the most important of which are phosphorus and nitrogen in available forms. In some cases, shortages of silicate can inhibit the growth of the diatoms and encourage growth of other species. In certain limited areas, other elements such as iron and manganese have been reported as limiting growth of algae, and the presence or absence of other growth stimulating substances, such as vitamin B_{12}, can influence both the amount and the character of plant species capable of growing. It should be noted that in the marine environment, several elements essential for plant growth such as potassium, magnesium, and sulfur, are present in great excess.

Organic material produced by natural phytoplankton populations produces an oxygen demand when the material is consumed or decomposed. Oxygen is produced by the process of photosynthesis, but this production occurs only near the surface during daylight when the amount of light penetrating the water is adequate. Due to the sedimentation of dead organic particulate material, decomposition usually takes place in the deep waters where photosynthetically produced oxygen is not available.

The amount of organic material which can be produced by marine phytoplankton as a result of the addition of fertilizing elements is dependent upon the composition of the organic material. Redfield et al. (1963)[420] give the following ratios as characteristic of living populations in the sea and of the changes which occur in amounts of various elements left in water as a result of algal growth

$$\begin{array}{rrrl}
\Delta O: & \Delta C: & \Delta N\ : & \Delta P = \\
276: & 106: & 16\ : & 1 \text{ by atoms or} \\
138: & 40: & 7\tfrac{1}{4}: & 1 \text{ by weight}
\end{array}$$

In addition to the readily available forms of phosphorus and nitrogen (dissolved orthophosphate, ammonia, nitrite, and nitrate), organic forms of phosphorus and nitrogen may be made available by bacterial decomposition. Some dissolved organic nitrogen compounds are also available for direct assimilation.

It should be emphasized that these ratios are not constant in the rigorous sense of the stoichiometric ratios in chemistry. The plant cells can both enjoy a "luxury" consumption of each element (Lund 1950)[414] or survive nutritional deficiencies (Ketchum 1939,[410] Ketchum et al. 1949[411]). In terms of the total production of organic material these variations are important only when concentrations of the elements are unusually low. It has been shown, for example, in New England coastal waters that nitrogen is almost completely removed from the sea water when there is still a considerable amount of phosphorus available in the system. Under these circumstances the plants will continue to assimilate phosphorus, even though total production of organic matter is limited by the nitrogen deficiency (Ketchum et al. 1958,[412] Ryther and Dunstan 1971[423]).

The amount of oxygen dissolved in sea water at equilibrium with the atmosphere is determined by salinity and temperature. Nutrient elements added to the marine environment should be limited so that oxygen content of the water is not decreased below the criteria given in the discussion of Dissolved Oxygen in this Section. In many polluted estuaries, the amount of fertilizing elements added in municipal sewage is sufficient to produce enough organic material to completely exhaust the oxygen supply during decomposition. The oxygen content of sea water and of fresh water at equilibrium with the atmosphere is presented for different temperatures in Table IV-9. For the purposes of this table, a sea water of 30 parts per thousand (‰) salinity has been used, which is characteristic of the nearshore coastal waters. The salinity effect on concentration of oxygen at saturation is minor compared to effects of temperature in the normal ranges found in coastal waters.

From the ratios of elements given above and the saturation values for oxygen, one can derive the effect of nutrient enrichment of marine waters. For example, from an addition of phosphorus and available nitrogen to final concentrations of 50 and 362.5 micrograms per liter respectively in the receiving water, enough organic material could be produced to remove 6.9 milligrams per liter of oxygen from the water. Data in Table IV-9 indicate that sea water with a salinity of 30 ‰ and a temperature of 25 C will contain, at saturation, 6.8 milligrams of oxygen per liter. This concentration of nutrients would thus permit the system to become anoxic and would violate the requirement that oxygen not be changed beyond levels expressed in the section on Dissolved Oxygen. Fresh water would contain 8.1 mg/l of oxygen at saturation at 25 C, so that the same amount of nutrient addition would remove 84 per cent of the available oxygen.

The example used might be considered to set an upper limit on the amount of these nutrients added to water. The actual situation is, of course, much more complicated. It is clear from the data in Table IV-9 that summer conditions place the most stringent restrictions on nutrient additions to the aquatic environment. Furthermore, the normal content of nutrients in the natural environment has to be considered. If these were already high, the amount of nutrients that could be added would have to be reduced. As mentioned above, the ratio of elements present in the natural environment would also be important. Nitrogen is frequently the element in minimum supply relative to the requirement of the phytoplankton, and addition of excess phosphorus under these circumstances has less influence than addition of nitrogen. Differences in the ratios of nitrogen to phosphorus may also modify the type of species present. Ryther (1954),[422] for example, found that unusually low nitrogen to phosphorus ratios in Moriches Bay and Great South Bay on Long Island, New York, encourage the growth of microscopic forms of *Nannochloris atomus* at the expense of the diatoms normally inhabiting this estuary.

TABLE IV-9—Effects of Salinity and Temperature on the Oxygen Content of Water in Equilibrium with Air at Atmospheric Pressure

Temperature C	Salinity ‰	Oxygen mg/l	Salinity ‰	Oxygen mg/l
25	30	6.8	0	8.1
20	30	7.4	0	8.9
10	30	9.1	0	10.9
0	30	11.65	0	14.15

Richards and Corwin 1956[421].

Many forms of blue-green algae are capable of fixing nitrogen from the gaseous nitrogen dissolved in sea water. Nitrogen deficiencies could be replenished by this mechanism so that decrease in phosphorus content without concomitant decrease in nitrogen content might still lead to overenrichment, as well as shift the dominant phytoplankton population.

Oxygen content of upper water layers can be increased by exchanges with the atmosphere. This process is proportional to the partial pressure of oxygen in the two systems so that the more oxygen deficient the water becomes, the more rapid is the rate of replacement of oxygen in the water by atmospheric oxygen. Finally, mixing and dilution of the contaminated water with adjacent bodies of water could make additional oxygen available. All of these variables must be considered in order to determine acceptable levels at which nutrients present in sewage can be added to an aquatic environment. In fact, many polluted estuaries already contain excessive amounts of these fertilizing elements as a result of pollution by municipal sewage.

The effects of ratios of elements discussed above have a very important bearing upon some of the methods of control. For example, the removal of phosphates alone from the sewage will have an effect upon the processes of overenrichment only if phosphorus is indeed the element limiting production of organic matter. When nitrogen is limiting, as it is in New England coastal waters according to Ryther and Dunstan (1971),[423] the replacement of phosphorus by nitrogen compounds, such as nitrilotriacetate (NTA) could be more damaging to the ecosystem than continued use of phosphate-based detergents.

Pathogenic Microorganisms

The fecal coliform index is the most widely used microbiological index of sanitary quality of an estuary. Fecal coliform indices represent a compromise between the ideal of direct determination of bacterial and viral pathogens in time-consuming laboratory procedures, and the indirect, less indicative but practical exigencies. Laboratory methods for quantitative enumeration of virus currently are being developed and their present status is one of promise, but more time is needed for their evaluation. Bacterial pathogen detection frequently requires special laboratory attention.

Virus, in general, may exhibit considerably longer survival times in water and shellfish as compared to fecal coliform bacteria. Under these circumstances a negative *E. coli* test can give a false impression of the absence of viral pathogens (Slanetz et al. 1965,[424] Metcalf and Stiles 1968[415]). Fecal coliform multiplication may possibly occur in polluted waters leading to further difficulties in interpreting sanitary quality.

Disinfection of waste water by chlorine is effective in removing most pathogenic bacteria but unpredictable in reducing the number of viruses. Differences in resistance of bacteria and virus to chlorination may result in the appearance of infectious virus in treated effluents devoid of bacteria. Failure to demonstrate the presence of viruses would be the best way to insure their absence, but such capability awaits development of methods adequate for quantitative enumeration of virus in water.

The pollution of estuaries with waste products has led to the contamination of shellfish with human pathogenic bacteria and viruses. Outbreaks of infectious hepatitis and acute gastroenteritis derived from polluted shellfish have reinforced concern over the dangers to public health associated with the pollution of shellfish waters. The seriousness of viral hepatitis as a world problem has been documented by Mosley and Kendrick (1969).[416] Transmission of infectious hepatitis as a consequence of sewage-polluted estuaries has occurred through consumption of virus-containing shellfish, either raw or improperly cooked. Nine outbreaks of infectious hepatitis have been attributed to shellfish (Liu 1970).[413] Contamination of water by sewage leads to the closing of oyster beds to commercial harvesting, denying public use of a natural resource and causing economic repercussions in the shellfish industry. (See the discussion of Shellfish in Section I on Recreation and Aesthetics.)

Sludge Disposal into Marine Waters

Dumping of sewage sludge in the ocean continues and this practice, although at present indispensable, constitutes a loss of one resource and potential danger for another. A study on the New York Bight sludge and spoil dumping area has shown that an accumulation of toxic metals and petroleum materials appear to have reduced the abundance of the benthic invertebrates that normally rework the sediments in a healthy bottom community (Pearce 1969).[419]

Deep Sea Dumping

Biological degradation of organic waste materials is generally affected by micro-biota and chemophysical environmental factors. The deep sea is increasingly considered for the disposal of organic waste materials. A recent study (Jannasch et al. 1971)[409] has shown that rates of bacterial activity in degrading organic materials was slowed by about two orders of magnitude at depths of 5,000 to 15,000 feet as compared to samples kept at equal temperatures (38 F) in the laboratory. Since (a) the disposal of organic wastes should be designed on the basis of rapid decomposition and recycling, and (b) there is no control of the processes following deep-sea disposal, this environment cannot be considered a suitable or safe dumping site.

Potential Beneficial Uses of Sewage

Light loads of either organic-rich raw sewage or nutrient-rich biological treatment (secondary) effluent increase biological productivity. Except for short-term data on increased fish and shellfish production, beneficial effects have rarely been sufficiently documented, but at the present time several active research programs are underway. Some degree of nutrient enrichment exists today in most estuaries close to centers of populations. These estuaries remain relatively productive and useful for fishing and recreation. Certain levels of ecosystem modification via organic and nutrient enrichment appear to be compatible with current water uses; however, subtle changes in ecosystems may be accompanied by later, more extensive change.

The possibility of intensive use of essential plant nutrients in waste material to increase the harvestable productivity of estuarine coastal systems has been suggested as a logical way to treat sewage and simultaneously derive an economic benefit. Aquaculture systems would essentially be an extension of the waste treatment process. Conceptually, aquaculture is a form of advanced treatment. The limiting factor involves problems presented by toxic synthetic chemicals, petroleum, metals, and pathogenic microorganisms in effluents of conventional biological treatment plants.

Rationale for Establishing Recommendations

It is conceptually difficult to propose a level of nutrient enrichment that will not alter the natural flora because seasonal phytoplankton blooms with complex patterns of species succession are an integral part of the ecology of estuarine and coastal waters. The timing and intensity of blooms vary from year to year and patterns of species succession are frequently different in successive years. The highly productive and variable ecology of estuaries makes it difficult to differentiate between the early symptoms of artificial nutrient enrichment and natural cyclic phenomena. In addition, there have already been major quantitative and qualitative changes in the flora of marine waters close to centers of population. These changes are superimposed on the normal patterns of growth and may not in themselves impair the recreational and commercial use of waters.

Simulation modeling has been used to predict the total phytoplankton response to given nutrient inputs with success by O'Connor (1965)[418] and DiToro et al. (1971)[405] in the San Joaquin Estuary and by Dugdale and Whitledge (1970)[406] for an ocean outfall. Their models predict the phytoplankton response from the interaction of the kind and rate of nutrient loading and the hydrodynamic dispersal rates. This technique, although not perfect, facilitates evaluation of the ecological impact of given nutrient loads, but does not help in deciding what degree of artificial enrichment is safe or acceptable.

Recommendations

- **Untreated or treated municipal sewage discharges should be recognized as a major source of toxic substances. Recommendations for these constituents will limit the amount of sewage effluent that can be dispersed into estuaries. Reduced degradation rates of highly dispersed materials** should be considered if the effluent contains refractory organic material. Undegradable synthetic organic compounds do not cause oxygen depletion but can still adversely affect the ecosystem. Maintenance of dissolved oxygen standards will not prevent the potentially harmful buildup of these materials. Specific quantitative analyses should be done to identify and assess the abundance of these compounds.

- The addition of any organic waste to the marine environment should be carefully controlled to avoid decomposition which would reduce the oxygen content of the water below the levels specified in the recommendations for oxygen.
- Neither organic matter nor fertilizers should be added that will induce the production of organic matter by normal biota to an extent causing an increase in the size of any natural anoxic zone in the deeper waters of an estuary.
- The natural ratios of available nitrogen to total phosphorus should be evaluated under each condition, and the element actually limiting plant production should be determined. Control of the amount of the limiting element added to the water will generally control enrichment.
- If the maximum amounts of available nitrogen and phosphorus in domestic waste increase the concentration in receiving waters to levels of 50 micrograms per liter of phosphorus and 360 micrograms per liter of nitrogen, enough organic matter would be produced to exhaust the oxygen content of the water, at the warmest time of the year under conditions of poor circulation, to levels below those recommended (see p. 275). These concentrations of nutrients are clearly excessive.
- The potential presence of pathogenic bacteria and viruses must be considered in waters receiving untreated or treated municipal sewage effluents. The present quality standards for fecal coliform counts (see pp. 31 32) should be observed. The procedures for the examination of seawater and shellfish as recommended by Hosty et al. (1970)[illegible] should be used.
- Disposal of sludge into coastal waters may adversely affect aquatic organisms, especially the bottom fauna. Periodic examination samples should determine the spread of such an operation to aid in the control of local waste material loads. The probable transport by currents should be carefully considered. The dumping of sludge into marine waters should be recognized as a temporary practice.
- Disposal of organic wastes into the deep-sea is not recommended until further studies on their fate, their effect on the deep-sea fauna, and the controllability of such a procedure have been completed.

SOLID WASTES, PARTICULATE MATTER, AND OCEAN DUMPING

Disposal of solid wastes has become one of the most urgent and difficult problems in crowded urban centers. Ocean disposal of these waste materials is receiving increased attention as land suitable for disposal becomes increasingly difficult to find.

Solid wastes are of many types and each may have a different impact on the marine environment. Household and commercial rubbish as well as automobiles and sewage sludge are disposed of at sea. Industrial wastes may be either solid or dissolved material, of varying toxicity. Harbor channels need continuous dredging, temporarily increasing the suspended sediment load, and the spoils often are dumped in coastal waters. Building rubble and stone also often are placed in the sea. The impact of disposal of these different materials into the ocean will range from innocuous to seriously damaging.

Particulate material is also discharged to the ocean by surface runoff, sewage outfalls, and storm sewers (Municipality of Metropolitan Seattle 1965).[464] Much of this material settles to the bottom at or near the discharge site (Gross 1970).[443] An increasingly important method of disposal is that of barging solids offshore to be dumped in coastal areas. Table IV-10 shows compilation of the amounts of wastes barged to sea in 1968 on the Pacific, Atlantic, and Gulf Coasts (Smith and Brown 1969).[476]

Dredge Spoils

Dredge spoils make up a major share of sea disposal operations. Their composition depends upon the source from which they were obtained. Saila et al. (1968)[472] were able to differentiate between dredged spoil from Providence Harbor dumped offshore and sediments of the natural bottom in the dumping area (Rhode Island Sound). Gross (1970)[443] suggests that dredge spoil generally consists of a mixture of sands, silts, and wastes which form the surface deposits in harbors. He compared minor element concen-

TABLE IV-10—Ocean Dumping: Types and Amounts, 1968

Waste type	(In tons) Atlantic	Gulf	Pacific	Total
Dredge spoils	15,808,000	15,300,000	7,320,000	38,428,000
Industrial wastes	3,013,200	696,000	981,300	4,690,500
Sewage sludge	4,477,000	0	0	4,477,000
Construction and demolition debris	574,000	0	0	574,000
Solid waste	0	0	26,000	26,000
Explosives	15,200	0	0	15,200
Total	23,887,400	15,966,000	8,327,300	48,210,700

Council on Environmental Quality 1970[434].

trations in harbor sediments, dredged wastes, and continental shelf sediments. The median values of observed concentrations were clearly different, although the ranges of concentrations overlapped.

The proportion of dredging spoils from polluted areas is illustrated in Table IV-11.

A variety of coastal engineering projects involve changes in suspended loads and sedimentation (Ippen 1966,[447] Wicker 1965[480]). Because important biotic communities may inhabit the sites selected for these projects, conflicts arise concerning navigational, recreational, fisheries, conservation, and municipal uses of the areas (Cronin et al. 1969).[436] Although our knowledge about the effects is limited and the literature is widely scattered, Copeland and Dickens (1969)[433] have attempted to construct a picture of how dredging affects estuarine ecosystems from information gathered in the upper Chesapeake Bay, Maryland, Redfish Bay, Texas, and an intracoastal canal in South Carolina.

The biological effects of suspended loads, sedimentation, dredging methods and spoil disposal may range from gross damage, such as habitat destruction and smothering, to more subtle effects under low but chronic conditions of sedimentation over long periods of exposure. The channelization, dumping of spoils, dredging, and filling in the Gulf Coast estuaries had destroyed roughly 200,000 acres of swamp, marsh, and bay bottom areas by 1968 (Chapman 1968,[432] Marshall 1968[459]).

Mixtures of clays, silts, fine sands, and organic matter, sometimes referred to as "faunally rich muddy sand," tend to support larger benthic populations than coarse clean unstable sands, gravels, or soft muds (Carriker 1967)[430] over or through which locomotion may be difficult (Yonge 1953).[482] Close relationships exist between the presence of organic matter, the mechanical nature of sediments, and infaunal feeding habits (Sanders 1956,[473] 1958,[474] McNulty et al. 1962,[461] Brett cited by Carriker 1967[430]).

Ten years after dredging Boca Ciega Bay invertebrate recolonization of canal sediments (92 per cent silt and clay; 3.4 per cent carbon) was negligible. None of 49 fish species caught in these canals (as compared to 80 species in undredged areas) was demersal, apparently because of the lack of benthic fish food organisms on or in the canal deposits (Taylor and Saloman 1968).[477] Breuer (1962)[429] noted that layers of dead oyster shell in South Bay corresponded to layers of deposited spoil from dredging and redredging of the Brownsville Ship Channel. He thought that this suggested destruction of South Bay oyster populations with each dredging operation.

TABLE IV-11—Estimated Polluted Dredge Spoils

	Total spoils (in tons)	Estimated percent of total polluted spoils[a]	Total polluted spoils (in tons)
Atlantic Coast	15,808,000	45	7,120,000
Gulf Coast	15,300,000	31	4,740,000
Pacific Coast	7,320,000	19	1,390,000
Total	38,428,000	34	13,250,000

[a] Estimates of polluted dredge spoils consider chlorine demand; BOD; COD; volatile solids; oil and grease; concentrations of phosphorous, nitrogen, and iron; silica content; and color and odor of the spoils.

Council on Environmental Quality 1970[431]

Pfitzenmeyer (1970)[470] and Flemer et al. (1967)[441] noted a 71 per cent reduction in average number of individuals and a marked reduction in diversity and biomass in a spoil area in upper Chesapeake Bay after dredging ceased. One and one half years after dredging, the number of individuals and species diversity of the spoil disposal area, but not in the channel, were the same as those of the surrounding area.

In lower Chesapeake Bay, Harrison et al. (1964)[444] observed a transitory effect of a dredging and spoil disposal operation on infauna. Resettlement of the dredged and disposal areas was very rapid by active migration and hydrodynamic distribution of juveniles.

Mock (1967)[463] noted that an unaltered shore in Clear Lake, Texas, produced 2.5 times more post larval and juvenile brown shrimp (*Penaeus aztecus*) and 14 times more post larval and juvenile white shrimp (*Penaeus setiferus*) than a similar bulkheaded shore. In a laboratory study using similar substrates, Williams' (1958)[481] data suggested that the type of substrate may exert its influence through its effect on available cover, although a contributing factor may be the different food content of the substrate.

Bayless (1968)[427] observed higher average hatches of striped bass eggs (*Morone saxatillis*) on coarse sand (58.9 per cent) and a plain plastic pan (60.3 per cent) than on silt-sand (21 per cent), silt-clay-sand (4 per cent) or muck detritus (none). These results tend to support Mansueti's (1962)[458] and Huet's (1965)[446] contention that deposition of suspended matter may interfere with or prevent fish reproduction by destruction of demersal eggs in upper estuarine areas.

Sewage Sludges

Sewage sludges contain about 5 per cent solids which consist of about 55 per cent organic matter, 45 per cent aluminosilicates, and tend to contain concentrations of some heavy metals at least ten times those of natural sediments (Gross 1970).[443]

Sewage sludge has been dumped off New York Harbor since 1924 in the same area. Studies by Pearce (1970a,[465] b)[466] show that the normal bottom populations in an area of about 10 square miles have been eliminated and that the benthic community has been altered over an area of approximately 20 square miles. Even the nematodes, unusually tolerant to pollution, are relatively scarce in the smaller area. In areas adjacent to the sewage sludge disposal area the sea clams have been found to be contaminated by enteric bacteria and the harvest of these clams in this area has been prohibited. The oxygen content of the water near the bottom is very low, less than 10 per cent of saturation in August, the warmest time of year. Chemical analysis of the sludge deposits have shown not only high organic content but also high concentrations of heavy metals and petrochemicals. In this area of the New York Bight, fin-rot disease of fish has been observed and is being investigated (Pearce 1970b).[466] In laboratory tests it has been shown that sludge deposits can cause necrosis of lobster (*Homarus americanus*) and crab shells and tend to clog their gills so that survival of these species in contact with the sludge deposits is very brief. In other laboratory experiments, organisms given a choice of substrate tend to avoid the sludge material in favor of the walls of the container or other surfaces that were made available (Pearce 1970b).[466] These studies have indicated that the disposal of sewage sludge has had disastrous ecological effects on the populations living on or near the bottom.

Many aspects associated with sludge dumping in the New York Bight require further investigation. It is not known, for example, how much of the material being dumped there is accumulating and how much is being decomposed. The effects of heavy metals, of oxygen-demanding materials, and of other components are imperfectly understood. When the rate of delivery of organic waste materials to an aquatic environment exceeds its capacity to recover, the rate of deterioration can be rapid. If, or when, sewage sludge disposal in this particular area of the New York Bight is terminated studies could determine whether the bottom populations can repopulate the area.

Solid Wastes

The amount of household and commercial rubbish to be disposed of in the United States is about 5 lbs per capita per day and is expected to increase to 7-½ lbs per capita per day (for a larger population), by the end of the present decade. Proposals have been made to collect and bale waste for transportation to the sea where it would be dumped in waters 1000 meters deep or more. It would be necessary that the bales be compacted to a density greater than sea water so that they would sink, and that no loose floating objects would be released from the bale. Among the suggestions made is that the bales be wrapped in plastic to avoid any leaching from the contents.

Pearce (1971)[468] reports that bales of compacted garbage wrapped in plastic and reinforced paper disintegrated in a few weeks when placed in water 10 to 20 meters deep off the coast of New Jersey. Compacted bales of refuse were also anchored at a depth of 200 meters off the Virgin Islands Pearce (1970c).[467] These were retrieved and inspected after approximately three months of exposure. Little growth had occurred on the surface of the bales, but some polychaete worms had penetrated the bales to a depth of 2–3 cm., and the material within the bale had decomposed to a limited extent. Relatively high counts of total coliform bacteria (96,000 Most Probable Number, MPN) and of fecal coliforms (1,300 MPN) were found in materials retrieved from the interior of the bales, indicating prolonged survival or growth of these nonmarine forms and suggesting a possible hazard of introduction of pathogens to the sea. The ecological effects of disposing of these materials are inadequately known.

Disposal of solid wastes, including dredging spoils and sewage sludge into the deep waters off the edge of the Continental Shelf (more than 200 meters) has been frequently suggested as a way to protect the inshore biota. However, the rate of decomposition of organic material at the high pressure and low temperature of the deep sea is very much slower than it would be at the same low temperature at atmospheric pressure (Jannasch et al. 1971).[449] The organisms in the deep sea have evolved in an extremely constant environment. They are, therefore, unaccustomed to the unusual stresses which confront organisms in more variable situations typical of coastal waters. Biologists interested in studying the bottom populations of the deep sea are extremely concerned about altering these populations before there is an opportunity to study them thoroughly.

Industrial Wastes

A wide variety of industrial waste is being dumped at sea. If this is discharged as a solution or slurry from a moving ship or barge it will be diluted in the turbulent wake and by the normal turbulence of the sea (Ford and Ketchum 1952).[442] The recommendations for mixing zones (p. 231) and for the constituents of specific waste material included should be applied to each such operation.

One such operation which has been extensively studied is the disposal of acid-iron wastes in the New York Bight (Redfield and Walford 1951,[471] Ketchum et al. 1951,[451] Vacarro et al. 1972,[478] Wiebe et al. *in press* 1972[479]). Even though this disposal has proceeded for over twenty years, no adverse effects on the marine biota have been demonstrated. The acid is rapidly neutralized by sea water and the iron is precipitated as nontoxic ferric hydroxide. This is a flocculant precipitate and the only accumulation above normal background levels in the sediments appears to be in the upper end of the Hudson Canyon, close to the specified dumping area. The so-called "acid grounds" have become a favored area among local blue fishermen. More toxic materials would clearly present an entirely different set of problems. This illustrates the need for a rational approach to problems of ocean dumping.

Other Solid Wastes

Automobiles are sometimes dumped at sea, and some work has been done on an experimental basis in an effort to determine whether artificial reefs can be created from them to improve sport fishing. There is evidence that the number of fish caught over these artificial reefs is greater than over a flat level bottom, but it is not yet certain whether this represents an aggregation of fishes already in the area or an actual increase in productivity.

Disposal of building rubble (brick, stone, and mortar) at sea is not widely practiced. Presumably, this material could form artificial reefs and attract populations of fish, both as a feeding ground and by providing some species with cover. Obviously, the bottom organisms present would be crushed or buried, but Pearce (1970a,[465] b)[466] found no permanent detrimental effects in the building rubble disposal site off New York City.

Suspended Particulate Materials

In addition to specific waste disposal operations, suspended particulate material, seston, may be derived from other sources, and have a variety of biological effects. Particulate material can originate from detritus carried by rivers, atmospheric fallout, biological activity, chemical reactions, and resuspension from the bottom as a result of currents, storms, or dredging operations. The particles introduced by rivers can be rock, mineral fragments, and clay serving as a substrate for microorganisms or affecting light transmission in the water column. In addition, organic matter fragments, which make up 20 to 40 per cent of particles in coastal waters (Biggs 1970,[428] Manheim et al. 1970[457]) may comprise 50 per cent to 80 per cent of suspended material further offshore. Particle concentrations generally range from 1 to 30 mg/l in coastal waters to about 0.1 to 1 mg/l at the surface in the open ocean. Higher concentrations occur near the bottom.

The estimated yearly sediment load from rivers to the world oceans is estimated at 20 to 36 × 10^8 tons with 80 per cent originating in Asia (Holeman 1968).[445] Much of this load is trapped in estuaries and held inshore by the general landward direction of subsurface coastal currents (Meade 1969).[462] Gross (1970)[443] suggests that 90 per cent or more of particles originating from rivers or discharged to the oceans settles out at the discharge site or never leaves the coastal zone.

Average seston values may more than double from natural causes during a tidal cycle. Biggs (1970)[428] observed concentrations in the upper Chesapeake Bay ranging from less than 20 mg/l to greater than 100 mg/l during a single day. Resuspension of bottom sediments by storm waves and currents induced by wind were responsible for this range of concentrations. Masch and Espey (1967)[460] found that the total suspended material concentrations in Galveston Bay, Texas, ranged from 72 mg/l in the surface water of the ship channel to over 150 g/l six inches above the bay bottom near dredging operations. Normal background concentrations in Galveston Bay during times of strong wind action were 200 to 400 mg/l. Background values observed by Mackin (1961)[456] in Louisiana marshes ranged from 20 to 200 mg/l. Depending on the amount of overburden, operation times, and rate of discharges, Masch and Espey (1967)[460] recorded suspended fixed solids concentrations in dredge discharges ranging from 3,000 to 29,100 mg/l.

The basic relationships between physical and chemical aspects of suspended and deposited sediments and the responses of estuarine and marine organisms are poorly understood (Sherk 1971).[475] However, there is general agreement that particulate material in suspension or settling on the bottom can affect aquatic organisms both directly and indirectly, by mortality or decreased yield.

Particles suspended in the water column can decrease light penetration by absorption and scattering and thus limit primary productivity. Resuspended sediments exert an oxygen demand on the order of eight times that of the same material in bottom deposits (Isaac 1965).[448] Jitts (1959)[450] found that 80 to 90 per cent of phosphate in solution was absorbed by silt suspensions which might also modify the rate of primary production. However, exchange rates and capacity of sediment can maintain a favorable level of phosphate (1 micromole/l) for plant production (Pomeroy et al. 1965).[469] Carritt and Goodgal (1954)[431] postulated a mechanism for phosphate removal, transport, and regeneration by the sediment-phosphate sorption complex at different temperatures, pH values, and salinities.

Evidence tends to support the contention that nutrient fertilization and possible release of toxic materials can occur with resuspension of bottom material in the water column (Gross 1970).[443] This may occur during dredging, disposal and dumping operations, reagitation during storms or floods and from beach erosion. In upper Chesapeake Bay total phosphate and nitrogen were observed to increase over ambient levels by factors of 50 to 1,000 near an overboard spoil disposal project, but no gross effects were observed in samples incubated with water from the spoil effluent (Flemer et al. 1967,[441] Flemer 1970[440]).

Oyster and clam eggs and larvae demonstrate a remarkable ability to tolerate the variable turbidities of the estuarine environment at concentrations up to 4.0 g/l (Carriker 1967,[430] Davis and Hidu 1969[438]). Survival and growth of these egg and larval stages reported by Davis (1960)[437] and Loosanoff (1962),[452] however, indicated a significant effect on survival at suspended particle concentrations of as little as 125 mg/l. Earlier life stages of the oyster tend to be more sensitive to lower concentrations of suspended material than adults. However, the effects on survival and growth cannot wholly be attributed to particle sizes and concentrations since different particle types may have markedly different effects at similar concentrations. The adult American oyster (*Crassostrea virginica*) appears to be a remarkably silt-tolerant organism when not directly smothered by deposited sediments (Lunz 1938,[454] 1942[455]). Significantly, mortality of adult oysters was not evident with suspended sediment concentrations as high as 700 mg/l (Mackin 1961),[456] but there was a drastic reduction in pumping rates (57 per cent at 100 mg/l of silt) observed by Loosanoff and Tommers (1948)[453] and Loosanoff

(1962[452]). Apparently adult oysters may pump at reduced rates throughout most of their lives when the background suspended particulate matter persists at values observed by Biggs (1970)[428] and Masch and Espey (1967[460]).

Organisms that colonize hard surfaces must contend with a sediment mat of varying thickness. While motile fauna may be able to adjust to short range vertical bottom alterations from scour or deposition, "... the capacity and behavior of less motile estuarine benthos in adjustment to relatively rapid fluctuations in the bottom level are little known. Fixed epifauna, like oysters and barnacles, perish when covered by sediment, adjustment occurring only indirectly through later repopulation of the area from elsewhere" (Carriker 1967).[430]

The highly variable nature of suspended loads (Biggs 1970),[428] the resuspension of bottom accumulations by currents, tidal action and wind, and the feeding and filtering activities of benthic organisms complicate the determination of threshold values or limiting conditions for aquatic organisms. Data are difficult to compare because of differences in methods and approaches. This may indicate a lack of understanding of sedimentation and the difficulty in distinguishing between the effect of light attenuation by suspended particles and the effects of these particles on growth and physiology of estuarine and marine organisms (Municipality of Metropolitan Seattle 1965).[461] The observed responses of organisms may not be due to turbidity or total suspended sediment concentration, but to the number of particles, their densities, sizes, shapes, types, presence and types of organic matter and the sorptive properties of the particles.

Physical alterations in estuaries and offshore dumping have had obvious effects on estuarine and marine biological resources. These effects have been given little consideration in project planning, however, and little information exists concerning the magnitude of biological change because few adequate studies have been attempted (Sherk 1971).[475] Areas of high biological value, such as nursery grounds or habitats for commercially important species, must be protected from sediment damage (Municipality of Metropolitan Seattle 1965).[461] For example, the exceptionally high value of the Upper Chesapeake as a low salinity fish nursery area has been demonstrated (Dovel 1970).[439] Larvae and eggs are particularly sensitive to environmental conditions, and sediment-producing activities in this type of area should be restricted to seasons or periods of least probable effects.

Results reported from the study of this area, concerning seasonal patterns of biota, the nature of the sediments, and physical hydrography of the area, can be applied to the other areas being considered for dredging, disposal, and dumping. These data, in addition to careful pre-decision surveys or research conducted at the site under consideration should provide a guide to efforts to minimize damage and enhance desirable features of the system (Cronin 1970).[435]

Adequate knowledge of local conditions at sites selected for any sediment-producing activity is essential, however. This will generally require preproject surveys for each site selected because knowledge of ecological impacts of these activities is limited. Data should be obtained on the "... biological values of the areas involved, seasonal patterns of the biota, the nature of the sediments, physical hydrography of the area, and the precise location of productive or potential shellfish beds, fish nursery areas and other areas of exceptional importance to human uses..." which are close to or in the site selected (Cronin 1970).[435]

Appropriate laboratory experiments are also required. These should have value in predicting effects of sedimentation in advance of dredging operations. Eventually, the results of these experiments and field observations should yield sets of environmental conditions and criteria, for adequate coastal zone management and competent guidance to preproject decision making (Sherk 1971).[475]

The presence of major benthic resources (e.g., oyster beds, clam beds) in or near the selected area should be cause for establishment of a safety zone or distance limit between them and the sediment-producing activity. This would control mortality caused by excessive deposition of suspended particulate material on the beds and prevent spread of spoil onto the beds from the disposal or dumping sites. Biggs (1970)[428] found that the maximum slope of deposited spoil was 1:100 and the average slope was 1:500 in the Upper Chesapeake. These slopes may prove useful in estimating safety zone limits on relatively flat bottoms. At times, the safety zone would have to be quite large. For example, the areas in New York Bight which are devoid of naturally occurring benthos in the sewage sludge and dredging spoil disposal areas were attributed to toxins, low dissolved oxygen, and the spreading of the deposits (Pearce 1970a).[465] The presence or absence of bottom currents or density flows should be determined (Masch and Espey 1967).[460] If these are present, measures must be taken to prevent transport of deposits ashore or to areas of major benthic resources.

Tolerable suspended sediment levels or ranges should accommodate the most sensitive life stages of biologically important species. The present state of knowledge dictates that the critical organism must be selected for each site where environmental modification is proposed.

Recommendations

The disposal of waste materials at sea, or the transport of materials for the purpose of disposal at sea should be controlled. Such disposal should be permitted only when reasonable evidence is presented that the proposed disposal will not seriously damage the marine biota, interfere with fisheries operations or with other uses of the marine environment such as navigation and recreation, or

cause hazards to human health and welfare. The following guidelines are suggested:

- Disposal at sea of potentially hazardous materials such as highly radioactive material or agents of chemical or biological warfare should be avoided.
- Toxic wastes should not be discharged at sea in a way which would adversely affect the marine biota. The toxicity of such materials should be established by bioassay tests and the concentrations produced should conform to the conditions specified in the discussion of mixing zones (pp. 231–232).
- Disposal of materials containing settleable solids or substances that may precipitate out in quantities adversely affecting the biota should be avoided in estuarine or coastal waters.
- Solid waste disposal at sea should be avoided if floating material might accumulate in harbors or on the beaches or if such materials might accumulate on the bottom or in the water column in a manner that will deleteriously affect deep sea biota.

In connection with dredging operations or other physical modifications of harbors and estuaries which would increase the suspended sediment load, the following types of investigations should be undertaken:

- Evaluation of the range and types of particles to be resuspended and transported, where they will settle, and what substratum changes or modifications may be created by the proposed activities in both the dredged and the disposal areas.
- Determination of the biological activity of the water column, the sediment-water interface, and the substrate material to depths which contain burrowing organisms.
- Estimation of the potential release into the water column of sediments, those substances originally dissolved or complexed in the interstitial water of the sediments, and the beneficial or detrimental chemicals sorbed or otherwise associated with particles which may be released wholly or partially after resuspension.
- Establish the expected relationship between properties of the suspended load and the permanent resident species of the area and their ability to repopulate the area, and the transitory species which use the area only at certain seasons of the year.

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INTRODUCTION

Modern agriculture increasingly depends upon the quality of its water to achieve the fullest production of domestic plants and animals and satisfy general farmstead needs. The quality of its water is important to modern agriculture not only in determining the productivity of plants and animals, but also as it affects the health and welfare of the human farm population.

Irrigation is one of the largest consumers of water for agricultural use. Differences in crop sensitivity to salinity and toxic substances necessitate the need for evaluating water quality criteria for irrigational purposes. Polluted water can be detrimental to animal health and to the safety and value of agricultural products. Good water quality is an important factor in the health and comfort of rural families needing water for drinking, food preparation, bathing, and laundering.

Discussions of water quality requirements relate in turn to problems of pollution posed by urban, industrial, and agricultural wastes. Some naturally occuring constituents, present in surface and groundwaters, can also adversely affect agricultural uses of water. Among these substances are suspended solids, dissolved organic and inorganic substances, and living organisms such as toxic algae and organisms associated with food spoilage. Where undesirable natural or foreign substances interfere with optimum water use, management and treatment practices must be implemented. Often there are simple but effective things that a farmer or rancher can do to manage and improve the quality of his water supply. Although considerations of water supply management are important, such matters are beyond the scope of this section on Agricultural Uses of Water, which is restricted to the quality requirements of water for domestic and other farmstead uses, for livestock, and for irrigation of crops.

Farmsteads typically require water at point of use, of quality equivalent to that demanded by urban populations, particularly for household uses, washing and cooling produce, and production of milk. Water of such high quality is frequently not readily available to the farmstead and often can be obtained only through water treatment. In the near future, water treatment facilities may be a routine installation in any well-designed farmstead operation. It is not the purpose of this section to elaborate upon treatment alternatives, but satisfactory treatment possibilities do exist for producing from most raw water a supply that will satisfy the quality needed for most agricultural uses.

The task of evaluating criteria and developing recommendations is complicated by the need to consider numerous complex interactions. For example, it is not practical to discuss water quality criteria for irrigation without considering crop responses to climatic and soil factors and their interrelationships with water. Evaluation of water quality requirements for livestock drinking water is also complicated by interactions of such variables as the quantity of water consumed and an animal's sex, size, age, and diet. It should, therefore, be emphasized that evaluating criteria is a complex task, and that using the recommendations in this report made on the basis of those criteria must be guided by expert judgment.

GENERAL FARMSTEAD USES OF WATER

This section considers quality requirements of water for use by the human farm population and for other uses associated with agricultural operations exclusive of livestock production and crop irrigation. Included are water for household uses, drinking water, and water for preparing produce and milk for marketing. For these purposes finished water of quality at least comparable to that intended for urban users is required at point of use.

Farmers and ranchers usually do not have access to the large, well-controlled water supplies of most municipalities and typically must make the best use of available surface or groundwater supplies. But there are problems associated with the use of these waters, which often contain objectionable natural constituents. These may be classified as suspended solids, dissolved inorganic salts and minerals, dissolved organic constituents, and living organisms, all of which occur naturally and are not introduced by man or as a result of his activities.

Suspended solids are organic and inorganic particles found in water supplies. They include sand, which is commonly associated with well supplies, and silt and clay frequantly found in untreated surface waters. Dissolved inorganic salts and minerals are found in both surface and groundwaters. Most of these are soluble salts consisting of calcium, magnesium, and sodium with associated anions (i.e., carbonate, bicarbonate, sulfate, and chloride). Greatest concentrations are found in the waters of arid and semiarid regions and in brackish waters along the sea coasts. In some western rivers total dissolved solids exceed 5,000 milligrams per liter (mg/l), although many contain less than 2,000 mg/l (Livingstone 1963).[24]* Surface waters draining from areas high in organic materials such as swamps and bogs often contain dissolved organic constituents composed mainly of hydroxy-carboxylic acids (Lamar and Goerlitz 1966,[21] Lamar 1968[20]) that impart a yellow or brown color to the water. Coloration often ranges from 100 to 800 platinum cobalt units compared to the 15 recommended by the federal Drinking Water Standards (Environmental Protection Agency 1972[11]).† Living organisms in standing bodies of water that impart objectionable odors and tastes for human consumption include algae, diatoms, and protozoa.

Because these constituents even in a properly protected supply of raw water used on farmsteads cause water quality that does not satisfactorily approximate the quality of potable water, it may be necessary to resort to water treatment. The wide range of quality characteristics associated with raw agricultural water supplies is matched by a broad range of water treatment methods. Microbial contaminants such as pathogenic or food spoilage bacteria, often present in surface waters, indicate that treatment is required to produce suitable water supplies. Treatments available include the use of halogens or sodium hypochlorite (Bauman and Ludwig 1962,[5] Black et al. 1965,[7] Kjellander and Lund 1965,[17] Water Systems Council 1965–1966,[41] Oliver 1966,[30] Laubusch 1971[22]), ozone (O'Donovan 1965),[29] silver (Shaw 1966,[32] Behrman 1968[6]), ultraviolet sterilization (Kristoffersen 1958,[19] Huff et al. 1965[14]), and heat (Shaw 1966)[32]. Reviews of some of the problems associated with farmstead water supplies and possible methods of treatment are given by Wright (1956),[42] Davis (1960),[8] Malaney et al. (1962),[26] James (1965),[15] Water Systems Council (1965–1966),[41] Elms (1966),[10] Kabler and Kreissl (1966),[16] Stover (1966),[33] and Atherton (1970).[2] Farmers, however, should seek expert advice in selecting from various treatment alternatives in order to achieve the desired quality of finished water.

A troublesome aspect of water quality for general farmstead uses, particularly regarding the handling of produce and milk, involves nonpathogenic bacterial contaminants. Many such microorganisms including algae are found even in properly protected agricultural water supplies (Thomas 1949,[34] Walters 1964),[40] and various kinds contribute to problems of color, odor, taste, and to rapid spoilage of contaminated products (American Water Works Assoc. Committee on Tastes and Odors 1970,[1] Mackenthun and Keup 1970).[25] For example, offensive odors are often attributable to sulfate-reducing bacteria (Lewis 1965).[23] Victoreen (1969)[39] discussed water coloration problems caused by *Arthrobacter*, a species of soil bacteria. Growths of "iron bacteria" in pipes may result in slimy masses that clog pipes and produce undesirable flavors (Kabler and Kreissl 1966).[16] Ropy milk, i.e., milk that forms threads or viscous masses when poured or dipped, is a typical problem often attributable to contaminated water (Thomas 1949,[34] Davis 1960[8]).

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* Citations are listed at the end of the Section. They can be located alphabetically within subtopics or by their superscript numbers which run consecutively across subtopics for the entire Section.

† Throughout this report, all references to the federal Drinking Water Standards are to those published by the Environmental Protection Agency, 1972.[11]

Psychrophilic bacteria can affect the storage quality of milk and other food products (Davis 1960,[8] Malaney et al. 1962,[26] Ayres 1963,[4] Thomas et al. 1966).[36] Similarly, thermoduric microorganisms are a problem in some farmstead water supplies, since they can withstand milk pasteurization temperatures and lead to spoilage (Thomas 1949,[34] Davis 1960,[8] Malaney et al. 1962).[26] Numerical recommendations for permissible levels of these and other nonpathogenic organisms have little current usefulness, because approximately 170 species of bacteria are known to occur in raw water supplies, and only half of them are observed during routine bacteriological examinations (Thomas 1949,[34] Malaney et al. 1962[26]). Similarly minimal contamination of perishable raw food materials with small residues of rinse water or splash can result in rapid growth under suitable temperature conditions to cause early spoilage of a high quality product.

Malaney et al. (1962)[26] stated that simple, commonly used water treatment processes render raw water supplies suitable for farmstead uses including handling of produce and milk.

WATER FOR HOUSEHOLD USES AND DRINKING

Every farm should have a dependable water supply that is palatable and safe for domestic use. This requirement dictates that the finished water be of quality comparable to that designated by the federal Drinking Water Standards for water supply systems used by interstate carriers and others subject to federal quarantine regulations. These standards have been found to be reasonable in terms of both the possibility of compliance and the acceptability of such water for domestic farmstead uses.

Groundwater sources are generally regarded as providing a more dependable supply and as being less variable in composition than surface water sources. However, many groundwater supplies contain excessive concentrations of soluble salts composed of calcium, magnesium, and associated anions (carbonate, bicarbonate, sulfate, and chloride), or hydrogen sulfide. They can cause taste, odor, acidity, and staining problems (Wright 1956,[42] Dougan 1966,[9] Kabler and Kreissl 1966,[16] Klumb 1966,[18] Behrman 1968[6]). In the ground waters of western states high concentrations of nitrates may occur. Levels may exceed the concentration of 10 mg/l of nitrate-nitrogen recommended by Section II on Public Water Supplies.

Because all supplies are subject to contamination, care must be exercised in both the installation and maintenance of water systems. Raw water should be free of impurities that are offensive to sight, smell, and taste (Wright 1956)[42] and free of significant concentrations of substances and organisms detrimental to public health (see Section II).

WATER FOR WASHING AND COOLING RAW FARM PRODUCTS

Many root crops, fruits, and vegetables are washed before leaving the farm for the market. Changes in fruit production associated with mechanical harvesting and bulk handling and an ever-increasing emphasis on quality have made the washing and hydrocooling of raw produce a common farm practice. Water for such uses should be of the same quality as that for drinking and household purposes, and as such should conform to Drinking Water Standards. It is important that water for processing raw produce be of good quality bacteriologically (Geldreich and Bordner 1971)[13] and free of substances imparting color, off-flavor, and off-odor (Mercer 1971).[27]

WATER FOR WASHING MILK-HANDLING EQUIPMENT AND COOLING DAIRY PRODUCTS

Water used to clean milk utensils may greatly affect the quality of milk (Atherton et al. 1962),[3] and since modern methods of milk production require large volumes of water, its quality must not be detrimental to milk. Steadily increasing demands for water due to intensified agricultural production have required many farm operators to develop secondary sources of water often of inferior quality (Esmay et al. 1955,[12] Pavelis and Gertel 1963).[31] Such supplies should be treated before use in milk-handling equipment (Thomas 1949,[34] Thomas et al. 1953[37]).

The Grade "A" Pasteurized Milk Ordinance of the United States Public Health Service (U.S. Department of Health, Education, and Welfare. Public Health Service 1965)[38] is accepted as the basic sanitation standard for raw milk supplies. Farm water supplies may meet these potable standards yet have a detrimental effect on the quality of modern milk supply. Rinse waters which are potable but contain psychrophylic microorganisms, excessive hardness, or iron or copper can have a very deleterious effect on dairy sanitation and milk quality unless properly treated to remove such contaminants (Davis 1960,[8] Atherton et al. 1962,[3] Atherton 1970,[2] Moore 1971[28]). The traditional concepts of potability and softness no longer suffice in this era of mechanized milk-handling systems. Lengthy storage of raw milk prior to pasturization and the possible breakdown of normal milk constituents by organisms able to grow at refrigeration temperatures may produce unacceptable changes in the quality of fluid milk or other manufactured dairy products (Thomas 1958,[35] Davis 1960,[8] Thomas et al. 1966[36]).

Water of quality comparable to that described in Drinking Water Standards typically suffices for the production of milk. However, it is important that the water at point of use be clear, colorless, palatable, free of harmful microorganisms, noncorrosive, and nonscale-forming (Moore 1971).[28]

Recommendations

For general farmstead uses of water, including drinking, other household uses, and handling of produce and milk, it is recommended that water of the quality designated by the federal Drinking Water Standards be used. Raw water supplies not meeting these requirements should be treated to yield a finished product of quality comparable to drinking water. In general, raw waters should be free of impurities that are offensive to sight, smell, and taste. At point of use, they should be free of significant concentrations of substances and organisms harmful to public health (see Section II: Public Water Supplies) and detrimental to the market value of agricultural products.

WATER FOR LIVESTOCK ENTERPRISES

Domestic animals represent an important segment of agriculture and are a vital source of food. Like man and many other life forms, they are affected by pollutants in their environment. This section is concerned primarily with considerations of livestock water quality and factors affecting it. These include the presence of ions causing excessive salinity, elements and ions which are toxic, biologically produced toxins, radionuclides, pesticide residues, and pathogenic and parasitic organisms.

Of importance in determining recommendations for these substances in livestock water supplies are the quantity of water an animal consumes per day and the concentration of the mineral elements in the water supply from which he consumes it. Water is universally needed and consumed by farm animals, but it does not account for their entire daily intake of a particular substance. Consequently, tolerance levels established for many substances in livestock feed do not accurately take into consideration the tolerance levels for those substances in water. Concentrations of nutrients and toxic substances in water affect an animal on the basis of the total amount consumed. Because of this, some assessment of the amounts of water consumed by live-stock on a daily basis and a knowledge of the probable quantity of elements in water and how they satisfy daily nutritional requirements are needed for determining possible toxicity levels.

WATER REQUIREMENTS FOR LIVESTOCK

The water content of animal bodies is relatively constant: 68 per cent to 72 per cent of the total weight on a fat-free basis. The level of water in the body usually cannot change appreciably without dire consequences to the animal; therefore, the minimal requirement for water is a reflection of water excreted from the body plus a component for growth in young animals (Robinson and McCance 1952,[53] Mitchell 1962[46]).

Water is excreted from the body in urine and feces, in evaporation from the lungs and skin, in sweat, and in productive secretions such as milk and eggs. Anything that influences any of these modes of water loss affects the minimal water requirement of the animal.

The urine contains the soluble products of metabolism that must be eliminated. The amount of urine excreted daily varies with the feed, work, external temperature, water consumption, and other factors. The hormone vasopressin (antidiuretic hormone) controls the amount of urine by affecting the reabsorption of water from the kidney tubules and ducts. Under conditions of water scarcity, an animal may concentrate its urine to some extent by reabsorbing a greater amount of water than usual, thereby lowering the animal's requirement for water. This capacity for concentration, however, is usually limited. If an animal consumes excess salt or a high protein diet, the excretion of urine is increased to eliminate the salt or the end products of protein metabolism, and the water requirement is thereby increased.

The amount of water lost in the feces varies depending upon diet and species. Cattle, for instance, excrete feces with a high moisture content while sheep, horses, and chickens excrete relatively dry feces. Substances in the diet that have a diuretic effect will increase water loss by this route.

Water lost by evaporation from the skin and lungs (insensible water loss) may account for a large part of the body's water loss approaching, and in some cases exceeding, that lost in the urine. If the environmental temperature is increased, the water lost by this route is also increased. Water lost through sweating may be considerable, especially in the case of horses, depending on the environmental temperature and the activity of the animal.

All these factors and their interrelation make a minimal water requirement difficult to assess. There is also the additional complication that a minimal water requirement does not have to be supplied entirely by drinking water. The animal has available to it the water contained in feeds, the metabolic water formed from the oxidation of nutrients, water liberated by polymerization, dehydration, or synthesis within the body, and preformed water associated with nutrients undergoing oxidation when the energy balance is negative. All of these may vary. The water available from the feed will vary with the kind of feed and with the amount consumed. The metabolic water formed from the oxidation of nutrients may be calculated by the use of factors obtained from equations of oxidation of typical proteins, fats, and carbohydrates. There are 41, 107, and 60 grams (g) of water formed per 100 g of protein, fat, and carbohydrate oxidized, respectively. In fasting animals, or those subsisting on a protein deficient diet, water may be formed from the destruction of tissue protein. In general, it is assumed that tissue protein is associated with three times its weight of water, so that per gram of tissue protein metabolized, three grams of water are released.

It has been found by careful water balance trials that the water requirement of various species is a function of body surface area rather than weight. This implies that the requirements are a function of energy metabolism, and Adolph (1933)[43] found that a convenient liberal standard of total water intake is 1 milliliter (ml) per calorie (cal) of heat produced. This method automatically included the increased requirement associated with activity. Cattle require somewhat higher amounts of water (1.29 to 2.05 g/cal) than other animals. However, when cattle's large excretion of water in the feces is taken into account, the values are approximately a gram per calorie.

For practical purposes, water requirements can be measured as the amount of water consumed voluntarily under specified conditions. This implies that thirst is a result of need.

Water Consumption of Animals

In dry roughage and concentrate feeding programs the water present in the feed is so small relative to the animal's needs that it may be ignored (Winchester and Morris 1956).[55]

Beef Cattle. Data calculated by Winchester and Morris (1956)[55] indicated that values for water intake vary widely depending primarily on ambient temperature and dry matter intake. European breeds consumed approximately 3.5, 5.3, 7.0, and 17 liters of water daily per kilogram (kg) dry matter ingested at 40, 70, 90, 100 F, respectively. Thus at an atmospheric temperature of 21 C (70 F), a 450 kg steer on a 9.4 kg daily dry matter ration would consume approximately 50 liters of water per day, while at 32 C (90 F) the expected daily water intake would be 66 liters.

Dairy Cattle. The calculations of Winchester and Morris (1956)[55] showed how water requirements varied with weight of cow, fat content of milk, ambient temperature, and amount needed per kilogram of milk daily. These investigations indicated that at 21 C (70 F) a cow weighing approximately 450 kg would consume about 4.5 liters of water per kilogram dry feed plus 2.7 l/kg of milk produced. Dairy heifers fed alfalfa and silage obtained about 20 per cent of their water requirements in the feed. Dairy cattle suffer more quickly from a lack of water than from a shortage of any other nutrient and will drink 3.0 to 4.0 kg of water per kilogram of dry matter consumed (National Research Council, Committee on Animal Nutrition, hereafter referred to as NRC 1971a).[52] Cows producing 40 kg of milk per day may drink up to 110 kg of water when fed dry feeds.

Sheep. Generally water consumption by sheep amounts to two times the weight of dry matter feed intake (NRC 1968b).[51] But many factors may alter this value, e.g., ambient temperature, activity, age, stage of production, plane of nutrition, composition of feed, and type of pasture. Ewes on dry feed in winter require four liters per head daily before lambing and six or more liters per day when nursing lambs (Morrison 1959).[48]

Swine. Pigs require 2 to 2.5 kg of water per kilogram of dry feed, but voluntary consumption may be as much as 4 to 4.5 kg in high ambient temperature (NRC 1968a).[50] Mount et al. (1971)[49] reported the mean water:feed ratios were between 2.1 and 2.7 at temperatures between 7 and 22 C, and between 2.8 and 5.0 at 30 and 33 C. The range of mean water consumption extended from 0.092 to 0.184 l/kg body weight per day. Leitch and Thomson (1944)[45] cited studies that demonstrated that a water-to-mash ratio of 3:1 gave the best results.

Horses. Leitch and Thomson (1944)[45] cited data that horses needed two to three liters of water per kg dry ration. Morrison (1936)[47] obtained data of a horse going at a trot that gave off 9.4 kg of water vapor. This amount was nearly twice that given off when walking with the same load, and more than three times as much as when resting during the same period.

Poultry. James and Wheeler (1949)[44] observed that more water was consumed by poultry when protein was increased in the diet; and more water was consumed with meat scrap, fish meal, and dried whey diets than with an all-plant diet. Poultry generally consumed 2 to 3 kg of water per kilogram of dry feed. Sunde (1967)[54] observed that when laying hens, at 67 percent production, were deprived of water for approximately 36 hours, production dropped to eight per cent within five days and did not return to the production of the controlled hens until 25–30 days later. Sunde (*personal communication* 1971)[56] prepared a table that showed that broilers increased on daily water consumption from 6.4 to 211 liters per 1,000 birds between two and 35 days of age, respectively. Corresponding water intake values for replacement pullets were 5.7 to 88.5 liters.

RELATION OF NUTRIENT ELEMENTS IN WATER TO TOTAL DIET

All the mineral elements essential as dietary nutrients occur to some extent in water (Shirley 1970).[66] Generally the elements are in solution, but some may be present in suspended materials. Lawrence (1968)[59] sampled the Chattahoochee River system at six different reservoirs and river and creek inlets and found about 1, 3, 22, 39, 61, and 68 per cent of the total calcium, magnesium, zinc, manganese, copper, and iron present in suspended materials, respectively. Any given water supply requires analysis if dietary decisions are to be most effective.

In the Systems for Technical Data (STORET) of the Water Programs Office of the Environmental Protection Agency, data (1971)[69] were accumulated from surface water analyses obtained in the United States during the period 1957–1969. These data included values for the mean, maximum, and minimum concentrations of the nutrient elements (see Table V-1). These values obviously include many samples from calcium-magnesium, sulfate-chloride and sodium-potassium, sulfate-chloride type of water as well as the more common calcium-magnesium, carbonate-bicarbonate type. For this reason the mean values for sodium, chloride, and sulfate may appear somewhat high.

Table V-2 gives the estimated average intake of drinking water of selected categories of various species of farm animals expressed as liters per day. Three values for each of calcium and salt are given for illustrative purposes. One column expresses the National Academy of Sciences value for daily requirement of the nutrient per day; the second gives the amount of the element contributed by the average concentration of the element (calculated from data in Table V-1) in the average quantity of water consumed daily; the third column gives the approximate percentage of the daily requirements contributed by the water drunk each day for each species of animal.

Magnesium, calculated as in Table V-2, was found to be present in quantities that would provide 4 to 11 per cent of the requirements for beef and dairy cattle, sheep, swine, horses, chickens, and turkeys.

Cobalt (Co) concentrations obtained by Durum et al. (1971)[58] were calculated, as they were more typical of water available to livestock than current values reported in STORET (1971).[69] A sufficient amount of Co was present at the median level to supply approximately three to 13 per cent of the dietary requirements of beef and dairy cattle, sheep, and horses. The NRC (1971a,[65] 1968b[61]) does not state what the cobalt requirements were for poultry and swine.

TABLE V-1—Water Composition, United States, 1957–69 (STORET) (Collected at 140 stations)

Substance	Mean	Maximum	Minimum	No Detns.
Phosphorus, mg/l	0.087	5.0	0.001	1,729
Calcium, mg/l	57.1	173.0	11.0	910
Magnesium, mg/l	14.3	137.0	8.5	1,143
Sodium, mg/l	55.1	7,500.0	0.2	1,001
Potassium, mg/l	4.3	370.0	0.06	1,004
Chloride, mg/l	478.0	19,000.0	0.000	37,355
Sulfate, mg/l	135.9	3,383.0	0.000	30,229
Copper, μg/l	13.8	280.0	0.8	1,071
Iron, μg/l	43.9	4,600.0	0.10	1,036
Manganese, μg/l	29.4	3,230.0	0.20	1,018
Zinc, μg/l	51.8	1,183.0	1.0	1,003
Selenium, μg/l	0.016	1.0	0.01	234
Iodine[a], μg/l	46.1	336.0	4.0	15
Cobalt[b], μg/l	1.0	5.0	0.000	720

[a] Dantzman and Breland (1970)[57].
[b] Durum et al. (1971)[58].

TABLE V-2—Daily Requirements of Average Concentrations of Calcium and Salt in Water for Various Animals

Animal	Daily[a] water intake, l	Calcium			Salt[d]		
		Required[b] daily gm	Average[c] amt. in drinking water, gm	Approx percentage of Req. in water	Required[b] daily gm	Amt. in[c] drinking water, gm	Percentage of Req. in water
Beef cattle 450 kg body wt.							
Nursing cow	60	28	3.4	12	25	8.5	34
Finishing steer	60	21	3.4	16	24	8.5	35
Dairy cattle 450 kg body wt.							
Lactating cow	90	76	5.1	7	66	12.7	19
Growing heifer	60	15	3.4	22	21	8.5	40
Maintenance, cow	60	12	3.4	28	21	8.5	40
Sheep							
Lactating ewe, 64 kg	6	6.8	0.3	5	13	0.9	7
Fattening lamb, 45 kg	4	3.1	0.2	7	10	0.6	6
Swine							
Growing, 30 kg	6	10.2	0.34	3	4.3	0.84	20
Fattening, 60 to 100 kg	8	16.5	0.46	3	4.3	1.12	26
Lactating sows, 200–250 kg	14	33.0	0.80	2	28.0	1.96	7
Horses 450 kg body wt							
Medium work	40	14	2.3	16	90	5.6	6
Lactating	50	30	2.9	10	90	7.1	8
Poultry							
Chickens, 8 weeks old	0.2	1.0	0.011	1	0.38	0.03	8
Laying hen	0.2	3.4	0.011	<1	0.44	0.03	6
Turkey	0.2	1.2	0.011	1	0.38	0.03	8

[a] See discussion on Water Consumption in text for sources of these values.
[b] Sources of values are the National Academy of Sciences, NRC Bulletins on Nutrient requirements.
[c] Calculated from Table 1.
[d] Based on sodium in water.

Sulfur values demonstrated that approximately 29 per cent of beef cattle requirements were met at average concentrations; dairy cattle 21 to 45 per cent; sheep 10 to 11 per cent; and horses 18 to 23 per cent of their requirements. The NRC (1971a,[65] 1968b[61]) do not give sulfur requirements for poultry and swine.

Iodine was not among the elements in the STORET accumulation, but values obtained by Dantzman and Breland (1970)[57] for 15 rivers and lakes in Florida can be used as illustrative values. Iodine was present in sufficient amounts to exceed the requirements of beef cattle and nonlactating horses and to meet 8 to 10 per cent of the requirements of sheep and 24 to 26 per cent of those of hens. Phosphorus, potassium, copper, iron, zinc, manganese, and selenium, when present at mean concentrations (Table V-1), would supply daily only one to four per cent or less of that recommended by the NRC (1966,[60] 1968a,[61] 1968b,[62] 1970,[63] 1971a,[64] 1971b[65]) for beef and dairy cattle, sheep, swine, horses, and poultry at normal water consumption levels.

If the maximum values shown in Table V-1 are present, some water would contain the dietary requirements of some species in the case of sodium chloride, sulfur, and iodine. Appreciable amounts of calcium, copper, cobalt, iron, manganese, zinc, and selenium would be present, if water were supplied with the maximum levels present. On the other hand, if the water has only the minimum concentration of any of the elements present, it would supply very little of the daily requirements.

It is generally believed that elements in water solution are available to the animal that consumes the water, at least as much as when present in solid feeds or dry salt mixes. This was indicated when Shirley et al. (1951,[67] 1957[68]) found that P^{32} and Ca^{45}, dissolved in aqueous solution as salts and administered as a drench, were absorbed at equivalent levels to the isotopes, when they were incorporated in forage as fertilizer and fed to steers, respectively. Many isotope studies have demonstrated that minerals in water consumed by animals are readily absorbed, deposited in their tissues, and excreted.

EFFECT OF SALINITY ON LIVESTOCK

It is well known that excessively saline waters can cause physiological upset or death of livestock. The ions most commonly involved in causing excessive salinity are calcium, magnesium, sodium, sulfate, bicarbonate, and chloride. Others may contribute significantly in unusual situations, and these may also exert specific toxicities separate from the osmotic effects of excessive salinity. (See Toxic Elements and Ions below.)

Early in this century, Larsen and Bailey (1913)[80] reported that a natural water varying from 4,546 to 7,369 mg/l of total salts, with sodium and sulfate ions predominating, caused mild diarrhea but no symptoms of toxicity in dairy cattle over a two-year period. Later, Ramsay (1924)[91] reported from his observations that cattle could thrive on water containing 11,400 mg/l of total salts, that they could live under certain conditions on water containing 17,120 mg/l, and that horses thrived on water with 5,720 mg/l and were sustained when not worked too hard on water with 9,140 mg/l.

The first extensive studies of saline water effects on rats and on livestock were made in Oklahoma (Heller and Larwood 1930,[76] Heller 1932,[74] 1933).[75] Rats were fed waters of various sodium chloride concentrations, and it was found among other things that (a) water consumption increased with salt concentration but only to a point after which the animals finally refused to drink until thirst drove them to it, at which time they drank a large amount at one time and then died; (b) older animals were more resistant to the effects of the salt than were the young; (c) the effects of salinity were osmotic rather than related to any specific ion; (d) reproduction and lactation were affected before growth effects were noted; (e) there appeared, in time, to be a physiological adjustment to saline waters; and (f) 15,000–17,000 mg/l of total salts seemed the maximum that could be tolerated, some adverse effects being noted at concentrations lower than this. With laying hens, 10,000 mg/l of sodium chloride in the drinking water greatly delayed the onset of egg production, but 15,000 mg/l or more were required to affect growth over a 10-week period. In swine, 15,000 mg/l of sodium chloride in the drinking water caused death in the smaller animals, some leg stiffness in the larger, but 10,000 mg/l did not appear particularly injurious once they became accustomed to it. Sheep existed on water containing 25,000 mg/l of sodium or calcium chloride or 30,000 mg/l of magnesium sulfate but not without some deleterious effects. Cattle were somewhat less resistant, and it was concluded that 10,000 mg/l of total salts should be considered the upper limit under which their maintenance could be expected. A lower limit was suggested for lactating animals. It was further observed that the animals would not drink highly saline solutions if water of low salt content was available, and that animals showing effects of saline waters returned quickly to normal when allowed a water of low salt content.

Frens (1946)[72] reported that 10,000 mg/l of sodium chloride in the drinking water of dairy cattle produced no symptoms of toxicity, while 15,000 mg/l caused a loss of appetite, decreased milk production, and increased water consumption with symptoms of salt poisoning in 12 days.

In studies with beef heifers, Embry et al. (1959)[71] reported that the addition of 10,000 mg/l of sodium sulfate to the drinking water caused severe reduction in its consumption, loss of weight, and symptoms of dehydration. Either 4,000 or 7,000 mg/l of added sodium sulfate increased water intake but had no effect on rate of gain or general health. Similar observations were made using waters with added sodium chloride or a mixture of salts, except that symptoms of dehydration were noted, and the mixed salts caused no increase in water consumption. Levels of up to 6,300 mg/l of added mixed salts increased water consumption in weanling pigs, but no harmful effects were observed over a three-month period.

In Australia, Peirce (1957,[83] 1959,[84] 1960,[85] 1962,[86] 1963,[87] 1966,[88] 1968a,[89] 1968b[90]) conducted a number of experiments on the salt tolerance of Merino wethers. Only minor harmful effects were observed in these sheep when they were confined to waters containing 13,000 mg/l or less of various salt mixtures.

Nevada workers have reported several studies on the effects of saline waters on beef heifers. They found that 20,000 mg/l of sodium chloride caused severe anorexia, weight loss, anhydremia, collapse, and certain other symptoms, while 10,000 mg/l had no effects over a 30-day period other than to increase water consumption and decrease blood urea (Weeth et al. 1960).[97] Additional experiments (Weeth and Haverland 1961)[98] again showed 10,000 mg/l to cause no symptoms of toxicity; while at 12,000 mg/l adverse effects were noted, and these intensified with increasing salt concentration in the drinking water. At a concentration of 15,000 mg/l, sodium chloride increased the ratio of urine excretion to water intake (Weeth and

Lesperance 1965),[100] and a prompt and distinct diuresis occurred when the heifers consumed water containing 5,000 or 6,000 mg/l following water deprivation (Weeth et al. 1968).[101] While with waters containing about 5,000 mg/l (Weeth and Hunter 1971)[99] or even less (Weeth and Capps 1971)[95] of sodium sulfate no specific ion effects were noted, heifers drank less, lost weight, and had increased methemoglobin and sulfhemoglobin levels. A later study (Weeth and Capps 1972)[96] gave similar results, but in addition suggested that the sulfate ion itself, at concentrations as low as 2150 mg/l had adverse effects.

In addition to the Oklahoma work, several studies on the effects of saline water on poultry have been reported. Selye (1943)[93] found that chicks 19 days old when placed on experiment had diarrhea, edema, weakness, and respiratory problems during the first 10 days on water containing 9,000 mg/l of sodium chloride. Later, the edema disappeared, but nephrosclerotic changes were noted. Water containing 3,000 mg/l of sodium chloride was not toxic to four-week-old chicks.

Others (Kare and Biely 1948)[77] observed that with two-day-old chicks on water containing 9,000 mg/l of added sodium chloride there were a few deaths, some edema, and certain other symptoms of toxicity. A solution with 18,000 mg/l of the salt was not toxic; however, when replaced on alternate days by fresh water, neither was it readily consumed.

Scrivner (1946)[92] found that sodium chloride in the drinking water of day-old poults at a concentration of 5,000 mg/l caused death and varying degrees of edema and ascites in over half of the birds in about two weeks. Sodium bicarbonate at a concentration of 1,000 mg/l was not toxic, at 3,000 mg/l caused some deaths and edema; and as the concentration increased above this, the effects were more pronounced. A solution containing 1,000 mg/l of sodium hydroxide caused death in two of 31 poults by 13 days, but the remainder survived without effects, and 7,500 mg/l of sodium citrate, iodide, carbonate, or sulfate each caused edema and many deaths.

South Dakota workers (Krista et al. 1961)[75] studied the effects of sodium chloride in water on laying hens, turkey poults, and ducklings. At 4,000 mg/l, the salt caused some increased water consumption, watery droppings, decreased feed consumption and growth, and increased mortality. These effects were more pronounced at a higher concentration, 10,000 mg/l, causing death in all of the turkey poults at two weeks, some symptoms of dehydration in the chicks, and decreased egg production in the hens. Experiments with laying hens restricted to water containing 10,000 mg/l of sodium or magnesium sulfate gave results similar to those for sodium chloride.

TABLE V-3—Guide to the Use of Saline Waters for Livestock and Poultry

Total soluble salts content of waters (mg/l)	Comment
Less than 1,000	Relatively low level of salinity. Excellent for all classes of livestock and poultry.
1,000–2,999	Very satisfactory for all classes of livestock and poultry. May cause temporary and mild diarrhea in livestock not accustomed to them or watery droppings in poultry.
3,000–4,999	Satisfactory for livestock, but may cause temporary diarrhea or be refused at first by animals not accustomed to them. Poor waters for poultry, often causing water feces, increased mortality, and decreased growth, especially in turkeys.
5,000–6,999	Can be used with reasonable safety for dairy and beef cattle, for sheep, swine, and horses. Avoid use for pregnant or lactating animals. Not acceptable for poultry.
7,000–10,000	Unfit for poultry and probably for swine. Considerable risk in using for pregnant or lactating cows, horses, or sheep, or for the young of these species. In general, use should be avoided although older ruminants, horses, poultry, and swine may subsist on them under certain conditions.
Over 10,000	Risks with these highly saline waters are so great that they cannot be recommended for use under any conditions.

In addition to the experimental work, there have been reports in the literature of field observations relating to the effects of excessively saline water (Ballantyne 1957,[70] Gastler and Olson 1957,[73] Spafford 1941[94]), and a number of guides to the use of these waters for livestock have been published (Ballantyne 1957,[70] Embry et al. 1959,[71] Krista et al. 1962,[79] McKee and Wolf, 1963,[81] Officers of the Department of Agriculture and the Government Chemical Laboratories 1950,[82] Spafford, 1941[94]). Table V-3 is based on the available published information. Among other things, the following items are suggested for consideration in using this table:

- Animals drink little, if any, highly saline water if water of low salt content is available to them.
- Unless they have been previously deprived of water, animals can consume moderate amounts of highly saline water for a few days without being harmed.
- Abrupt changes from water of low salinity to highly saline water cause more problems than a gradual change.
- Depressed water intake is very likely to be accompanied by depressed feed intake.

Table V-3 was developed because in arid or semiarid regions the use of highly saline waters may often be necessary. It has built into it a very small margin of safety, and its use probably does not eliminate all risk of economic loss.

Criteria for desirability of a livestock water are a somewhat different matter. These should probably be such that the risk of economic loss from using the water for any species or age of animals, lactating or not, on any normal feeding program, and regardless of climatic conditions, is almost nonexistent. On the other hand, they should be made no more severe than necessary to insure this small risk.

Recommendation

From the standpoint of salinity and its osmotic effects, waters containing 3,000 mg of soluble salts per liter or less should be satisfactory for livestock under almost any circumstance. While some minor physiological upset resulting from waters with

salinities near this limit may be observed, economic losses or serious physiological disturbances should rarely, if ever, result from their use.

TOXIC SUBSTANCES IN LIVESTOCK WATERS

There are many substances dissolved or suspended in waters that may be toxic. These include inorganic elements and their salts, certain organic wastes from man's activities, pathogens and parasitic organisms, herbicide and pesticide residues, some biologically produced toxins, and radionuclides.

For any of the above, the concentrations at which they render a water undesirable for use for livestock is subject to a number of variables. These include age, sex, species, and physiological state of the animals; water intake, diet and its composition, the chemical form of any toxic element present, and the temperature of the environment. Naturally, if feeds and waters both contain a toxic substance, this must be taken into account. Both short and long term effects and interactions with other ions or compounds must also be considered.

The development of recommendations for safe concentrations of toxic substances in water for livestock is extremely difficult. Careful attention must be given to the discussion that follows as well as the recommendations and to any additional experimental findings that may develop. Based on available research, an appropriate margin of safety, under almost all conditions, of specific toxic substances harmful to livestock that drink the waters and to man who consumes the livestock or their products, is reviewed below. Although the margin of safety recommended is usually large, the criteria suggested cannot be used as a guide in diagnosing livestock losses, since they are well below toxic levels for domestic animals.

Toxic Elements and Ions

Those ions largely responsible for salinity in water (sodium, calcium, magnesium, chloride, sulfate, and bicarbonate) are in themselves not very toxic. There are, however, a number of others that occur naturally or as the result of man's activities at troublesome concentrations. If feeds and water both contain a toxic ion, both must be considered. Interactions with other ions, if known, must be taken into account. Elements or ions become objectionable in water when they are at levels toxic to animals, where they seriously reduce the palatability of the water, or when they accumulate excessively in tissues or body fluids, rendering the meat, milk, eggs, or other edible product unsafe or unfit for human use.

Aluminum

Soluble aluminum has been found in surface waters of the United States in amounts to 3 mg/l, but its occurrence at such concentrations is rare because it readily precipitates as the hydroxide (Kopp and Kroner 1970).[192]

Most edible grasses contain about 15–20 mg/kg of the element. However, there is no evidence that it is essential for animal growth, and very little is found deposited in animal tissues (Underwood 1971).[254] It is not highly toxic (McKee and Wolf 1963,[193] Underwood 1971),[254] but Deobald and Elvehjem (1935)[138] found that a level of 4,000 mg aluminum per kilogram of diet caused phosphorus deficiency in chicks. Its occurrence in water should not cause problems for livestock, except under unusual conditions and with acid waters.

Recommendation

Livestock should be protected where natural drinking waters contain no more than 5 mg/l aluminum.

Arsenic

Arsenic has long been notorious as a poison. Nevertheless, it is present in all living tissues in the inorganic and in certain organic forms. It has also been used medicinally. It is accepted as a safe feed additive for certain domestic animals. It has not been shown to be a required nutrient for animals, possibly because its ubiquity has precluded the compounding of deficient diets (Frost 1967).[149]

The toxicity of arsenic can depend on its chemical form, its inorganic oxides being considerably more toxic than organic forms occurring in living tissues or used as feed additives. Differences in toxicities of the various forms are clearly related to the rate of their excretion, the least toxic being the most rapidly eliminated (Frost 1967,[149] Underwood 1971).[254] Except in unusual cases, this element should occur in waters largely as inorganic oxides. In waters carrying or in contact with natural colloidal material, the soluble arsenic content may be decreased to a very low level by adsorption.

Wadsworth (1952)[260] gave the acute toxicity of inorganic arsenic for farm animals as follows: poultry, 0.05–0.10 g per animal; swine, 0.5–1.0 g per animal; sheep, goats, and horses, 10.0–15.0 g per animal; and cattle, 15–30 g per animal. Franke and Moxon (1936)[145] concluded that the minimum dose required to kill 75 per cent of rats given intraperitoneal injections of arsenate was 14–18 mg arsenic per kilogram, while for arsenite it was 4.25–4.75 mg/kg of body weight.

When mice were given drinking water containing 5 mg/l of arsenic as arsenite from weaning to natural death, there was some accumulation of the element in the tissues of several organs, a somewhat shortened life span, but no carcinogenic effect (Schroeder and Balassa 1967).[233] In a similar study with rats (Schroeder et al. 1968b),[236] neither toxicity nor carcinogenic effects were observed, but large amounts accumulated in the tissues.

Peoples (1964)[220] fed arsenic acid at levels up to 1.25 mg/kg of body weight per day for eight weeks to lactating cows. This is equivalent to an intake of 60 liters of water containing 5.5 mg/l of arsenic (10.4 mg of arsenic acid) daily by a 500 kg animal. His results indicated that this form of arsenic is absorbed and rapidly excreted in the urine. Thus there was little tissue storage of the element; at no level of the added arsenic was there an increased arsenic content of the milk, and no toxicity was observed.

According to Frost (1967),[149] there is no evidence that 10 parts per million (ppm) of arsenic in the diet is toxic to any animal.

Arsenicals have been accused of being carcinogenic. This matter has been thoroughly reviewed by Frost (1967),[149] who concluded that they appear remarkably free of this property.

Most human foods contain less than 0.5 ppm of arsenic, but certain marine animals used as human food may concentrate it and may contain over 100 ppm (Frost 1967,[149] Underwood 1971[254]). Permissible levels of the element in muscle meats is 0.5 ppm; in edible meat by-products, 1.0 ppm; and in eggs, 0.5 ppm (U.S. Dept. of Health, Education, and Welfare, Food and Drug Administration 1963,[255] 1964[256]). Federal Drinking Water Standards list 0.05 mg/l as the upper allowable limit to humans for arsenic, but McKee and Wolf (1963)[193] suggested 1.0 mg/l as the upper limit for livestock drinking water. The possible role of biological methylation in increasing the toxicity (Chemical Engineering News 1971)[126] suggested added caution, however, and natural waters seldom contain more than 0.2 mg/l (Durum et al. 1971).[141]

Recommendation

To provide the necessary caution, and in view of available data, an upper limit of 0.2 mg/l of arsenic in water is recommended.

Beryllium

Beryllium was found to occur in natural surface waters only at very low levels, usually below 1 μg/l (Kopp and Kroner 1970).[182] Conceivably, however, it could enter waters in effluents from certain metallurgical plants. Its salts are not highly toxic, laboratory rats having survived for two years on a diet that supplied the element at a level of about 18 mg/kg of body weight daily. Pomelee (1953)[223] calculated that a cow could drink almost 1,000 liters of water containing 6,000 mg/l without harm, if these data for rats are transposable to cattle. This type of extrapolation must, however, be used with caution, and the paucity of additional data on the toxicity of beryllium to livestock precludes recommending at this time a limit for its concentration in livestock waters.

Boron

The toxicity of boron, its occurrence in foods and feeds, and its role in animal nutrition have been reviewed by McClure (1949),[190] McKee and Wolf (1963),[193] and Underwood (1971).[254] Although essential for plants, there is no evidence that boron is required by animals. It has a relatively low order of toxicity. In the dairy cow, 16–20 g of boric acid per day for 40 days produced no ill effects (McKee and Wolf 1963).[193]

There is no evidence that boron accumulates to any great extent in body tissues. Apparently, most natural waters could be expected to contain concentrations well below the level of 5.0 mg/l. This was the maximum amount found in 1,546 samples of river and lake waters from various parts of the United States, the mean value being 0.1 mg/l (Kopp and Kroner 1970).[182] Ground waters could contain substantially more than this at certain places.

Recommendation

Experimental evidence concerning the toxicity of this element is meager. Therefore, to offer a large margin of safety, an upper limit of 5.0 mg/l of boron in livestock waters is recommended.

Cadmium

Cadmium (Cd) is normally found in natural waters at very low levels. A nationwide reconnaissance of surface waters of the United States (Durum et al. 1971)[141] revealed that of over 720 samples, about four per cent contained over 10 μg/l of this element, and the highest level was 110 μg/l. Ground water on Long Island, New York, contained 3.2 mg/l as the result of contamination by waste from the electroplating industry, and mine waters in Missouri contained 1,000 mg/l (McKee and Wolf 1963).[193]

Research to date suggests that cadmium is not an essential element. It is, on the other hand, quite toxic. Man has been sickened by about 15 ppm in popsicles, 67 ppm in punch, 300 ppm in a cold drink, 530 ppm in gelatin, and 14.5 mg taken orally; although a family of four whose drinking water was reported to contain 47 ppm had no history of ill effects (McKee and Wolf 1963).[193]

Extensive tests have been made on the effects of various levels of cadmium in the drinking water on rats and dogs (McKee and Wolf 1963).[193] Because of the accumlation and retention of the element in the liver and kidney, it was recommended that a limit of 100 μg/l, or preferably less, be used for drinking waters.

Parizek (1960)[219] found that a single dose of 4.5 mg Cd/kg of body weight produced permanent sterility in male rats. At a level of 5 mg/l in the drinking water of rats (Schroeder et al. 1963a)[238] or mice (Schroeder et al. 1963b),[239] reduced longevity was observed. Intravenous injection of cadmium sulfate into pregnant hamsters at a level of 2 mg Cd/kg of body weight on day eight of gestation caused malformations in the fetuses (Mulvihill et al. 1970).[200]

Miller (1971)[196] studied cadmium absorption and distribution in ruminants. He found that only a small part of ingested cadmium was absorbed, and that most of what was went to the kidneys and liver. Once absorbed, its turnover rate was very slow. The cow is very efficient in keeping cadmium out of its milk, and Miller concluded that most major animal products, including meat and milk, seemed quite well protected against cadmium accumulation.

Interactions of cadmium with several other trace elements (Hill et al. 1963,[172] Gunn and Gould 1967,[159] Mason and Young 1967)[189] somewhat confuse the matter of establishing criteria.

Recommendation

From the available data on the occurrence of cadmium in natural waters, its toxicity, and its accumulation in body tissues, an upper limit of 50 μg/l allows an adequate margin of safety for livestock and is recommended.

Chromium

In a five-year survey of lake and river waters of the United States (Kopp and Kroner 1970),[182] the highest level found in over 1,500 samples was about 0.1 mg/l, the average being about 0.001 mg/l. In another similar survey (Durum et al. 1971)[141] of 700 samples, none contained over 0.05 mg/l of chromium VI and only 11 contained more than 0.005 mg/l. A number of industrial processes however use the element, which then may be discharged as waste into surface waters, possibly at rather high levels.

Even in its most soluble forms, the element is not readily absorbed by animals, being largely excreted in the feces; and it does not appear to concentrate in any particular mammalian tissue or to increase in these tissues with age (Mertz 1967,[194] Underwood 1971[254]).

Hexavalent chromium is generally considered more toxic than the trivalent form (Mertz 1967).[194] However, in their review of this element, McKee and Wolf (1963)[193] suggested that it has a rather low order of toxicity. Further, Gross and Heller (1946)[158] found that for rats the maximum nontoxic level, based on growth, for chromium VI in the drinking water was 500 mg/l. They also found that this concentration of the element in the water did not affect feed utilization by rabbits. Romoser et al. (1961)[226] found that 100 ppm of chromium VI in chick diets had no effect on the performance of the birds over a 21-day period.

In a series of experiments, Schroeder et al. (1963a,[238] 1963b,[239] 1964,[234] 1965[235]) administered water containing 5 mg/l of chromium III to rats and mice on low-chromium diets over a life span. At this level, the element was not toxic, but instead it had some beneficial effects. Tissue levels did not increase significantly with age.

As a result of their review of chromium toxicity, McKee and Wolf (1963)[193] suggested that up to 5 mg/l of chromium III or VI in livestock drinking water should not be harmful. While this may be reasonable, it may be unnecessarily high when the usual concentrations of the element in natural waters is considered.

Recommendation

An upper allowable limit of 1.0 mg/l for livestock drinking waters is recommended. This provides a suitable margin of safety.

Cobalt

In a recent survey of surface waters in the United States (Durum et al. 1971)[141] 63 per cent of over 720 samples were found to contain less than 0.001 mg/l of cobalt. One sample contained 4.5 mg/l, one contained 0.11 mg/l, and three contained 0.05–0.10 mg/l.

Underwood (1971)[254] reviewed the role of cobalt in animal nutrition. This element is part of the vitamin B_{12} molecule, and as such it is an essential nutrient. Ruminants synthesized their own vitamin B_{12} if they were given oral cobalt. For cattle and sheep a diet containing about 0.1 ppm of the element seemed nutritionally adequate. A wide margin of safety existed between the required and toxic levels for sheep and cattle, which were levels of 100 times those usually found in adequate diets being well tolerated.

Nonruminants required preformed vitamin B_{12}. When administered to these animals in amounts well beyond those present in foods and feeds, cobalt induced polycythemia (Underwood 1971).[254] This was also true in calves prior to rumen development; about 1.1 mg of the element per kg of body weight administered daily caused depression of appetite and loss of weight.

Cobalt toxicity was also summarized by McKee and Wolf (1963).[193]

Recommendation

In view of the data available on the occurrence and toxicity of cobalt, an upper limit for cobalt in livestock waters of 1.0 mg/l offers a satisfactory margin of safety, and should be met by most natural waters.

Copper

The examination of over 1,500 river and lake waters in the United States (Kopp and Kroner 1970)[182] yielded, at the highest, 0.28 mg/l of copper and an average value of 0.015 mg/l. These rather low values were probably due in part to the relative insolubility of the copper ion in alkaline medium and to its ready adsorbability on colloids (McKee and Wolf 1963).[193] Where higher values than those reported above are found, pollution from industrial sources or mines can be suspected.

Copper is an essential trace element. The requirement for chicks and turkey poults from zero to eight weeks of age is 4 ppm in the diet (NRC 1971b).[206] For beef cattle on rations low in molybdenum and sulfur, 4 ppm in the diet is adequate; but when these elements are high, the copper requirement is doubled or tripled (NRC 1970).[204] A dietary level of 5 ppm in the forage is suggested for pregnant and lactating ewes and their lambs (NRC 1968b[203]). A level of 6 ppm in the diet is considered adequate for swine (NRC 1968a).[202]

Swine are apparently very tolerant of high levels of copper, and 250 ppm or more in the diet have been used to improve liveweight gains and feed efficiency (Nutrition Reviews 1966a[210]; NRC 1968a).[202] On the other hand, sheep were very susceptible to copper poisoning (Underwood 1971),[254] and for these animals a diet containing 25 ppm was considered toxic. About 9 mg per animal per day was considered the safe tolerance level (NRC 1968b).[203]

Several reviews of copper requirements and toxicity have been presented (McKee and Wolf 1963,[193] Nutrition Reviews 1966a,[210] Underwood 1971).[254] There is very little experimental data on the effects of copper in the water supply on animals, and its toxicity must be judged largely from the results of trials where copper was fed. The element does not appear to accumulate at excessive levels in muscle tissues, and it is very readily eliminated once its administration is stopped. While most livestock tolerate rather high levels, sheep do not (NRC 1968b).[203]

Recommendation

It is recommended that the upper limit for copper in livestock waters be 0.5 mg/l. Very few natural waters should fail to meet this.

Fluorine

The role of fluorine as a nutrient and as a toxin has been thoroughly reviewed by Underwood (1971).[254] (Unless otherwise indicated, the following discussion, exclusive of the recommendation, is based upon this review.) While there is no doubt that dietary fluoride in appropriate amounts improved the caries resistance of teeth, the element has not yet been found essential to animals. If it is a dietary essential, its requirement must be very low. Its ubiquity probably insures a continuously adequate intake by animals.

Chronic fluoride poisoning of livestock has, on the other hand, been observed in several areas of the world, resulting in some cases from the consumption of waters of high fluoride content. These waters come from wells in rock from which the element has been leached, and they often contain 10–15 mg/l. Surface waters, on the other hand, usually contain considerably less than 1 mg/l.

Concentrations of 30–50 ppm of fluoride in the total ration of dairy cows is considered the upper safe limit, higher values being suggested for other animals (NRC 1971a).[205] Maximum levels of the element in waters that are tolerated by livestock are difficult to define from available experimental work. The species, volume, and continuity of water consumption, other dietary fluoride, and age of the animals, all have an effect. It appears, however, that as little as 2 mg/l may cause tooth mottling under some circumstances. At least a several-fold increase in its concentration seems, however, required to produce other injurious effects.

Fluoride from waters apparently does not accumulate in soft tissues to a significant degree. It is transferred to a very small extent into the milk and to a somewhat greater degree into eggs.

McKee and Wolf (1963)[193] have also reviewed the matter of livestock poisoning by fluoride, concluding that 1.0 mg/l of the element in their drinking water did not harm these animals. Other more recent reports presented data suggesting that even considerably higher concentrations of fluoride in the water may, with the exception of tooth mottling, caused no animal health problems (Harris et al. 1963,[166] Shupe et al. 1964,[246] Nutrition Reviews 1966b,[211] Saville 1967,[231] Schroeder et al. 1968a[237]).

Recommendation

An upper limit for fluorides in livestock drinking waters of 2.0 mg/l is recommended. Although this level may result in some tooth mottling it should not be excessive from the standpoint of animal health or the deposition of the element in meat, milk, or eggs.

Iron

It is well known that iron (Fe) is essential to animal life. Further, it has a low order of toxicity. Deobald and Elvehjem (1935)[138] found that iron salts added at a level of 9,000 mg Fe/kg of diet caused a phosphorus deficiency in chicks. This could be overcome by adding phosphate to the diet. Campbell (1961)[124] found that soluble iron salt administered to baby pigs by stomach tube at a level of 600 mg Fe/kg of body weight caused death within six hours. O'Donovan et al. (1963)[212] found very high levels of iron in the diet (4,000 and 5,000 mg/kg) to cause phosphorus deficiency and to be toxic to weanling pigs. Lower levels (3,000 mg/kg) apparently were not toxic. The intake of water by livestock may be inhibited by high levels of this element (Taylor 1935).[250] However, this should not be a common or a serious problem. While iron occurs in natural waters as ferrous salts which are very soluble, on contact with air it is oxidized and it precipitates as ferric oxide, rendering it essentially harmless to animal health.

It is not considered necessary to set an upper limit of acceptability for iron in water. It should be noted, however, that even a few parts per million of iron can cause clogging of lines to stock watering equipment or an undesirable staining and deposit on the equipment itself.

Lead

Lake and river waters of the United States usually contain less than 0.05 mg/l of lead (Pb), although concentrations in excess of this have been reported (Durum et al. 1971,[141] Kopp and Kroner 1970).[182] Some natural waters in areas where galena is found have had as much as 0.8 mg/l of the element. It may also be introduced into waters in the effluents from various industries, as the result of action of the water on lead pipes (McKee and Wolf 1963),[193] or by deposition from polluted air (NRC 1972).[207]

A nutritional need for lead by animals has not been demonstrated, but its toxicity is well known. A rather complete review of the matter of lead poisoning by McKee and Wolf (1963)[193] suggested that for livestock the toxicity of the element had not been clearly established from a quantitative standpoint. Even with more recent data (Donawick 1966,[139] Link and Pensinger 1966,[186] Harbourne et al. 1968,[165] Damron et al. 1969,[131] Hatch and Funnell 1969,[168] Egan and O'Cuill 1970,[143] Aronson 1971),[108] it is difficult to establish clearly at what level of intake lead becomes toxic, although a daily intake of 6–7 mg Pb/kg of body weight has been suggested as the minimum that eventually gave rise to signs of poisoning in cattle (Hammond and Aronson 1964).[164] Apparently, cattle and sheep are considerably more resistant to lead toxicosis than are horses, being remarkably tolerant to the continuous intake of relatively large amounts of the element (Hammond and Aronson 1964,[164] Garner 1967,[152] Aronson 1971[108]; NRC 1972[207]). However, there is some tendency for it to accumulate in tissues and to be transferred to the milk at levels that could be toxic to man (Hammond and Aronson 1964).[164]

There is some agreement that 0.5 mg/l of lead in the drinking water of livestock is a safe level (McKee and Wolf 1963);[193] and the findings of Schroeder and his associates with laboratory animals are in agreement with this (1963a,[238] 1963b,[239] 1964,[234] 1965[235]). Using 10 times this level, or 5 mg/l, of lead in the drinking water of rats and mice over their life spans, these authors observed no obvious direct toxic effects but did find an increase in death rates in the older animals, especially in the males. Schroeder et al. (1965)[235], observed that the increased mortality was not caused by overt lead poisoning, but rather by an increased susceptibility to spontaneous infections. Hemphill et al. (1971)[171] later reported that mice treated with subclinical doses of lead nitrate were more susceptible to challenge with *Salmonella typhimurium*.

Recommendation

In view of the lack of information concerning the chronic toxicity of lead, its apparent role in reducing disease resistance, and the very low incidence in natural waters of lead contents exceeding the 0.05 mg/l level, an upper limit of 0.1 mg/l for lead in livestock waters is recommended.

Manganese

Like iron, manganese is a required trace element, occurs in natural waters at only low levels as manganous salts, and is precipitated in the presence of air as manganic oxide. While it can be toxic when administered in the feed at high levels (Underwood 1971),[254] it is improbable that it would be found at toxic levels in waters.

It is doubtful that setting an upper limit of acceptability is necessary for manganese, but as with iron, a few milligrams per liter in water can cause objectionable deposits on stock watering equipment.

Mercury

Natural waters may contain mercury originating from the activities of man or from naturally occurring geological stores (Wershaw 1970,[262] White et al. 1970).[263] The element tends to sorb readily on a variety of materials, including the bottom sediments of streams, greatly reducing the levels that might otherwise remain in solution (Hem 1970).[170] Thus, surface waters in the United States have usually been found to contain much less than 5 μg/l of mercury (Durum et al. 1971).[141] In areas harboring mercury deposits, their biological methylation occurs in bottom sediments (Jensen and Jernelöv 1969)[176] resulting in a continuous presence of the element in solution (Greeson 1970).[156]

In comparison to the relative instability of organic compounds such as salts of phenyl mercury and methoxyethyl mercury (Gage and Swan 1961,[151] Miller et al. 1961,[195] Daniel and Gage 1969,[132] Daniel et al. 1971[133]) alkyl mercury compounds including methyl mercury (CH_3Hg^+) have a high degree of stability in the body (Gage 1964,[150] Miller et al. 1961)[195] resulting in an accumulative effect. This relative stability, together with efficient absorption from the gut, contributes to the somewhat greater toxicity of orally administered methyl mercury as compared to poorly absorbed inorganic mercury salts (Swensson et al. 1959).[249]

The biological half-life of methyl mercury varies from about 20 to 70 days in most species (Bergrund and Berlin 1969).[113] Brain, liver, and kidney were the organs that accumulated the highest levels of the element, with the distribution of methyl and other alkyl mercury compounds favoring nerve tissue and inorganic mercury favoring the kidney (Malishevskaya et al. 1966,[188] Platonow 1968,[222] Aberg et al. 1969).[102]

Transfer of methyl mercury (Curley et al. 1971),[130] but not mercuric mercury (Berlin and Ullberg 1963),[114] to the fetus has been observed. The element also appeared in the eggs of poultry (Kiwimae et al. 1969)[180] and wild birds (Borg et al. 1969,[118] Dustman et al. 1970)[142] but did not seem to concentrate there much above levels found in the tissues of the adult. Data concerning levels of mercury that may be detrimental to hatchability of eggs are too meager to support conclusions at this time. Also, data concerning transfer of mercury to milk is lacking.

The animal organs representing the principal tissues for mercury concentration are brain, liver, and kidney. It is desirable that the maximum allowable limit for mercury in livestock waters should result in less than 0.5 ppm of accumulated mercury in these tissues. This is the level now in use as the maximum allowable in fish used for human consumption.

Few data are available quantitatively relating dietary mercury levels with accumulation in animal tissues. The ratios between blood and brain levels of methyl mercury appeared to range from 10 for rats to 0.2 for monkeys and dogs (International Committee on Maximum Allowable Concentrations of Mercury Compounds 1969).[174] In addition, blood levels of mercury appeared to increase approximately in proportion to increases in dietary intake (Birke et al. 1967[115]; Tejning 1967[251]).

Assuming a 0.2 or more blood-to-tissue (brain or other tissue) ratio for mercury in livestock, the maintenance of less than 0.5 ppm mercury in all tissues necessitates maintaining blood mercury levels below 0.1 ppm. This would indicate a maximum daily intake of 2.3 μg of mercury per kilogram body weight. Based upon daily water consumption by meat animals in the range of up to about eight per cent of body weight, it is estimated that water may contain almost 30 μg/l of mercury as methyl mercury without the limits of these criteria being exceeded. Support for this approximation was provided in part by the calculations of Aberg et al. (1969)[102] showing that after "infinite" time the body burden of mercury in man will approximate 15.2 times the weekly intake of methyl mercury. Applying these data to meat animals consuming water equivalent to eight per cent of body weight and containing 30 μg/l of mercury would result in an average of 0.25 ppm mercury in the whole animal body.

Recommendation

Until specific data become available for the various species, adherence to an upper limit of 10 μg/l of mercury in water for livestock is recommended, and this limit provides an adequate margin of safety to humans who will subsequently not be exposed to as much as 0.5 ppm of mercury through the consumption of animal tissue.

Molybdenum

Underwood (1971)[254] reviewed the matter of molybdenum's role in animal nutrition. While the evidence that it is an essential element is good, the amount of molybdenum required has not been established. For cattle, for instance, no minimum requirement has been set, but it is believed to be low, possibly less than 0.01 ppm of the dry diet (NRC 1970).[204]

McKee and Wolf (1963)[193] reviewed the matter of toxicity of molybdenum to animals, but Underwood (1971)[254] pointed out that many of the studies on its toxicity are of limited value because a number of factors known to influence its metabolism were not taken into account in making these studies. These factors included the chemical form of molybdenum, the copper status and intake of the animal, the form and amount of sulfur in the diet, and other less well defined matters. In spite of these, there are data to support real species differences in terms of tolerance to the element. Cattle seem the least tolerant, sheep a little more so, and horses and swine considerably more tolerant.

While Shirley et al. (1950)[245] found that drenching steers daily with sodium molybdate in an amount equialent to about 200 ppm of molybdenum in the diet for a period of seven months resulted in no marked symptoms of toxicity, cattle on pastures where the herbage contained 20–100 ppm of molybdenum on a dry basis developed a toxicosis known as teart. Copper additions to the diet have been used to control this (Underwood 1971).[254]

Cox et al. (1960)[127] reported that rats fed diets containing 500 and 800 ppm of added molybdenum showed toxicity symptoms and had increased levels of the element in their livers. Some effects of the molybdenum in the diets on liver enzymes in the rats were not observed in calves that had been maintained on diets containing up to 400 ppm of the element.

Apparently, natural surface waters very rarely contained levels of this element of over 1 mg/l (Kopp and Kroner 1970),[182] which seemed to offer no problem.

Conclusion

Because there are many factors influencing toxicity of molybdenum, setting an upper allowable limit for its concentration in livestock waters is not possible at this time.

Nitrates and Nitrites

Livestock poisoning by nitrates or nitrites is dependent upon the intake of these ions from all sources. Thus, water or forage may independently or together contain levels that are toxic. Of the two, nitrite is considerably more toxic. Usually it is formed through the biological reduction of nitrate in the rumen of cattle or sheep, in freshly chopped forage, in moistened feeds, or in waters contaminated with organic matter to the extent that they are capable of supporting microbial growth. While natural waters often contain high levels of nitrate, their nitrite content is usually very low.

While some nitrate was transferred to the milk, Davison and his associates (1964)[135] found that for dairy cattle fed 150 mg NO_3N/kg of body weight the milk contained about 3 ppm of NO_3N. They concluded that nitrates in cattle feeds did not appear to constitute a hazard to human health, and that animals fed nitrate continuously developed some degree of adaptation to it.

The LD50 of nitrate nitrogen for ruminants was found to be about 75 mg NO_3N/kg of body weight when administered as a drench (Bradley et al. 1940)[119] and about 255 mg/kg of body weight when sprayed on forage and feed (Crawford and Kennedy 1960).[128] Levels of 60 mg NO_3N/kg of body weight as a drench (Sapiro et al. 1949)[230] and 150 mg NO_3N/kg of body weight in the diet (Prewitt and Merilan 1958;[224] Davison et al. 1964[135]) had no deleterious effects. Lewis (1951)[184] found that 60 per cent conversion of hemoglobin to methemoglobin occurred in mature sheep from 4.0 g of NO_3N or 2.0 g of NO_2N placed in the rumen, or 0.4 g NO_2N injected intravenously. As an oral drench, 90 mg NO_3N/kg of body weight gave peak methemoglobin levels of 5–6 g/100 ml of blood in sheep, while intravenous injection of 6 mg NO_2N/kg of body weight gave similar results (Emerick et al. 1965).[144]

Nitrate-induced abortions in cattle and sheep have generally required amounts approaching lethal levels (Simon et al. 1959,[247] Davison et al. 1962,[136] Winter and Hokanson 1964,[266] Davison et al. 1965[137]).

Some experiments have demonstrated reductions in plasma or liver vitamin A values resulting from the feeding of nitrate to ruminants (Jordan et al. 1961,[178] Goodrich et al. 1964,[153] Newland and Deans 1964,[209] Hoar et al. 1968[173]). The destructive effect of nitrites on carotene (Olson et al. 1963[213]) and vitamin A (Pugh and Garner 1963[225]) under acid conditions that existed in silage or in the gastric stomach have also been noted. On the other hand, nitrate levels of about 0.15 per cent in the feed (equivalent to about 1 per cent of potassium nitrate) have not been shown to influence liver vitamin A levels (Hale et al. 1962,[161] Weichenthal et al. 1963,[261] Mitchell et al. 1967[197]) nor to have other deleterious effects in controlled experiments, except for a possible slight decrease in production.

Assuming a maximum water consumption in dairy cattle of 3 to 4 times the dry matter intake (NRC 1971a[205]), the concentration of nitrate to be tolerated in the water should be about one-fourth of that tolerated in the feed. This would be about 300 mg/l of NO_3N.

Gwatkin and Plummer (1946)[160] drenched pigs with potassium nitrate solutions, finding that it required in excess of 300 mg NO_3N/kg of body weight to cause erosion and hemorrhage of the gastric mucosa and subsequent death. Lower levels of this salt had no effect when administered daily for 30 days. Losses in swine due to methoglobinemia have occurred only with the consumption of preformed nitrite and not with nitrate (McIntosh et al. 1943,[192] Gwatkin and Plummer 1946,[160] Winks et al. 1950[265]). Nitrate administered orally as a single dose was found to be acutely toxic at 13 mg NO_2N/kg of body weight, 8.7 mg/kg of body weight producing moderate methemoglobinemia (Winks et al. 1950).[265] Emerick et al. (1965)[144] produced moderate methemoglobinemia in pigs with intravenous injections of 6.0 mg NO_2N/kg of body weight and found that the animals under one week of age were no more susceptible to poisoning than older ones.

Drinking water containing 330 mg/l NO_3N fed continuously to growing pigs and to gilts from weaning through two farrowing seasons had no adverse effects (Seerley et al. 1965).[242] Further, 100 mg/l of NO_2N in drinking water had no effect on performance or liver vitamin A values of pigs over a 105-day experimental period, and methemoglobin values remained low. This level of nitrite greatly exceeded the maximum of 13 mg/l NO_2N found to form in waters in galvanized watering equipment and in the presence of considerable organic matter containing up to 300 mg/l NO_3N.

In special situations involving the presence of high levels of nitrates in aqueous slurries of plant or animal tissues, nitrite accumulation reached a peak of about one-fourth to one-half the initial nitrate concentration (McIntosh et al. 1943,[192] Winks et al. 1950,[265] Barnett 1952).[109] This situation was unusual, but since wet mixtures are sometimes used for swine, it must be considered in establishing criteria for water.

Levels of nitrate up to 300 mg/l NO_3N or of nitrite up to 200 mg/l of NO_2N were added to drinking waters without adverse effects on the growth of chicks or production of laying hens (Adams et al. 1966).[104] At 200 mg/l NO_2N, nitrite decreased growth in turkey poults and reduced the liver storage of vitamin A in chicks, laying hens, and turkeys. At 50 mg/l NO_2N, no effects were observed on any of the birds. Kienholz et al. (1966)[179] found that 150 mg/l of NO_3N in the drinking water or in the feed of chicks or poults had no detrimental effect on growth, feed efficiency, methemoglobin level, or thyroid weight, while Sell and Roberts (1963)[243] found that 0.12 per cent (1,200 ppm) of NO_2N in chick diets lowered vitamin A stores in the liver and caused hypertrophy of the thyroid. Other studies have shown poultry to tolerate levels of nitrate or nitrite similar to or greater than those mentioned above (Adams et al. 1967,[105] Crawford et al. 1969[129]). Up to 450 mg/l of NO_3N in the drinking water of turkeys did not significantly affect meat color (Mugler et al. 1970).[199]

Some have suggested that nitrate or nitrite can cause a chronic or subclinical toxicity (Simon et al. 1959,[247] McIlwain and Schipper 1963,[191] Pfander 1961,[221] Beeson 1964,[111] Case 1957[125]). Some degree of thyroid hypertrophy may occur in some species with the consumption of subtoxic levels of nitrate or nitrite (Bloomfield et al. 1961,[117] Sell and Roberts 1963),[243] but possibly not in all (Jainudeen et al. 1965).[175] In the human newborn, a chronic type of methemoglobinemia may result from feeding waters of low NO_3N content (Armstrong et al. 1958).[107] It appears, however, that all classes of livestock and poultry that have been studied under controlled experimental conditions can tolerate the continued ingestion of waters containing up to 300 mg/l of NO_3N or 100 mg/l of NO_2N.

Recommendation

In order to provide a reasonable margin of safety to allow for unusual situations such as extremely high water intake or nitrite formation in slurries, the NO_3N plus NO_2N content in drinking waters for livestock and poultry should be limited to 100 ppm or less, and the NO_2N content alone be limited to 10 ppm or less.

Selenium

Rosenfeld and Beath (1964)[227] have reviewed the problems of selenium poisoning in livestock. Of the three types of this poisoning described, the "alkali disease" syndrome required the lowest level of the element in the feed for its causation. Moxon (1937)[198] placed this level at about 5 ppm, and subsequent research confirmed this figure. Later work established that the toxicity of selenium was very similar when the element was fed as it occurs in plants, as selenomethionine or selenocystine, or as inorganic selenite or selenate (Halverson et al. 1962,[162] Rosenfeld and Beath 1964,[227] Halverson et al. 1966[163]). Ruminant animals may tolerate more as inorganic salts than do monogastric animals because of the salts' reduction to insoluble elemental form by rumen microorganisms (Butler and Peterson 1961).[121]

A study with rats (Schroeder 1967)[232] revealed that selenite, but not selenate, in the drinking water caused deaths at a level of 2 mg/l and was somewhat more toxic than selenite administered in the diet. However, the results of drenching studies with cattle and sheep (Maag and Glenn 1967)[187] indicated that selenium concentration in the water should be slight, if it is any more toxic in the same chemical form administered in the feed. If there are differences with respect to the effect of mode of ingestion on toxicity, they are probably small.

To date, no substantiated cases of selenium poisoning in livestock by waters have been reported, although some spring and irrigation waters have been found to contain over 1 mg/l of the element (Byers 1935,[122] Williams and Byers 1935,[261] Beath 1943[110]). As a rule, well, surface, and ocean waters appeared to contain less than 0.05 mg/l, usually considerably less. Byers et al. (1938)[124] explained the low selenium content as a result of precipitation of the selenite ion with ferric hydroxide. Microbial activity, however, removed either selenite or selenate from water (Abu-Erreish 1967);[103] this may be another explanation.

In addition to its toxicity, the essential role of selenium in animal nutrition (Thompson and Scott 1970)[252] must be considered. Between 0.1 and 0.2 ppm in the diet have been recommended as necessary to insure against a deficiency in poultry (Scott and Thompson 1969),[241] against white muscle disease in ruminants (Muth 1963),[201] and other diseases in other animals (Hartley and Grant 1961).[167] Selenium therapy suggests it as a requirement for livestock in general. Inorganic selenium was not incorporated into tissues to the same extent as it occurred in plant tissue (Halverson et al. 1962,[162] 1966,[163] Rosenfeld and Beath 1964[227]). It is doubtful that 0.2 ppm or less of added inorganic selenium appreciably increased the amount found in the tissue of animals ingesting it. The data of Kubota et al. (1967)[183] regarding the occurrence of selenium poisoning suggested that over a good part of the United States livestock were receiving as much as 0.5 ppm or even more of naturally occurring selenium in their diets continuously, without harm to them and without accumulating levels of the element in their tissues that make meats or livestock products unfit for human use.

Recommendation

It is recommended that the upper limit for selenium in livestock waters be 0.05 mg/l.

Vanadium

Vanadium has been present in surface waters in the United States in concentrations up to 0.3 mg/l, although most of the analyses showed less than 0.05 mg/l (Kopp and Kroner 1970).[182]

Recently, vanadium was determined essential for the growing rat, physiologically required levels appearing to be at or below 0.1 ppm of the diet (Schwarz and Milne 1971).[240] It became toxic to chicks when incorporated into the diet as ammonium metavanadate at concentrations over about 10 ppm of the element (Romoser et al. 1961,[226] Nelson et al. 1962,[208] Berg 1963,[112] Hathcock et al. 1964[169]). Schroeder and Balassa (1967)[233] found that when mice were allowed drinking water containing 5 mg/l of vanadium as vanadyl sulfate over a life span, no toxic effects were observed, but the element did accumulate to some extent in certain organs.

Recommendation

It is recommended that the upper limit for vanadium in drinking water for livestock be 0.1 mg/l.

Zinc

There are many opportunities for the contamination of waters by zinc. In some areas where it is mined, this metal has been found in natural waters in concentrations as high as 50 mg/l. It occurs in significant amounts in effluents from certain industries. Galvanized pipes and tanks may also contribute zinc to acidic waters. In a recent survey of surface waters, most contained less than 0.05 mg/l but some exceeded 5.0 mg/l, the highest value being 42 mg/l (Durum et al. 1971).[141]

Zinc is relatively nontoxic for animals. Swine have tolerated 1,000 ppm of dietary zinc (Grimmet et al. 1937,[157] Sampson et al. 1942,[229] Lewis et al. 1957,[185] Brink et al. 1959[120]), while 2,000 ppm or more have been found to be toxic (Brink et al. 1959).[120] Similar findings have been reported for poultry (Klussendorf and Pensack 1958,[181] Johnson et al. 1962,[177] Vohra and Kratzer 1968[250]) where zinc was added to the feed. Adding 2,320 mg/l of the element to water for chickens reduced water consumption, egg production, and body weight. After zinc withdrawal there were no symptoms of toxicity in chickens (Sturkie 1956).[248] In a number of studies with ruminants, Ott et al. (1966a,[215] b,[216] c,[217] d[218]) found zinc added to diets as the oxide to be toxic, but at levels over 500 mg/kg of diet.

While an increased zinc intake reflected an increase in level of the element in the body tissues, the tendency for its accumulation was not great (Drinker et al. 1927,[140] Thompson et al. 1927,[253] Sadasivan 1951,[228] Lewis et al. 1957),[185] and tissue levels fell rapidly after zinc dosing was stopped (Drinker et al. 1927,[140] Johnson et al. 1962[177]).

Zinc is a dietary requirement of all poultry and livestock. National Research Council recommendation for poults up to eight weeks was 70 mg/kg of diet; for chicks up to eight weeks, it was 50 mg/kg of diet (NRC 1971b);[206] for swine, 50 mg/kg of diet (NRC 1968a).[202] There is no established requirement for ruminants, but zinc deficiencies were reported in cattle grazing forage with zinc contents ranging between 18 and 83 ppm (Underwood 1971).[254] There is also no established requirement for sheep, but lambs fed a purified diet containing 3 ppm of the element developed symptoms of a deficiency that were prevented by adding 15 ppm of zinc to the diet; 30 ppm was required to give maximum growth (Ott et al. 1965).[214]

Cereal grains contained on the average 30–40 ppm and protein concentrates from 20 to over 100 ppm (Davis 1966).[134] In view of this, and in view of the low order of toxicity of zinc and its requirement by animals, a limit in livestock waters of 25 mg zinc/l would have a very large margin of safety. A higher limit does not seem necessary, since there would be few instances where natural waters would carry in excess of this.

Recommendation

It is recommended that the upper limit for zinc in livestock waters be 25 mg/l.

Toxic Algae

The term "water bloom" refers to heavy scums of blue-green algae that form on waters under certain conditions. Perhaps the first report of livestock poisoning by toxic algae was that of Francis (1878)[147] who described the problem in southern Australia. Fitch et al. (1934)[146] reviewed a number of cases of algal poisoning in farm animals in Minnesota between 1882 and 1933. All were associated with certain blue-green algae often concentrated by the wind at one end of the lake. Losses in cattle, sheep, and poultry were reported. The algae were found toxic to laboratory animals on ingestion or intraperitoneal injection.

According to Gorham (1964)[155] six species of blue-green algae have been incriminated, as follows:

Nodularia spumigena
Microsystis aeruginosa
Coelosphaerium Kuetzingianum
Gloeotrichia echinulata
Anabaena flos-aquae
Aphanizomenon flos-aquae

Of the above, Gorham states that *Microcystis* and *Anabaena* have most often been blamed for serious poisonings and algal blooms consisting of one or more of these species vary considerably in their toxicity (Gorham 1964).[155] According to Gorham (1960),[154] this variability seems to depend upon a number of factors, e.g., species and strains of algae that are predominant, types and numbers of bacterial associates, the conditions of growth, collection and decomposition, the degree of animal starvation and susceptibility, and the amount consumed. To date, only one toxin from blue-green algae has been isolated and identified, only from a few species and streams. This was a cyclic polypeptide containing 10 amino acid residues, one of which was the unnatural amino acid D-serine (Bishop et al. 1959).[116] This is also referred to as FDF (fast-death factor), since it causes death more quickly than SDF (slow-death factor) toxins produced in water blooms.

Shilo (1967)[244] pointed out that the sudden decomposition of algal blooms often preceded mass mortality of fish, and similar observations were made with livestock poisonings. This suggests that the lysis of the algae may be important in the release of the toxins, but it also suggests that in some circumstances botulism may be involved. The lack of oxygen may have caused the fish kill and must also be considered.

Predeath symptoms in livestock have not been carefully observed and described. Post-mortem examination is apparently of no help in diagnosis (Fitch et al. 1934).[146] Feeding or injecting algal suspensions or water from suspect waters have been used to some extent, but the occasional fleeting toxicity of these materials makes this procedure of limited value. Identification of any of the toxic blue-green algae species in suspect waters does no more than suggest the possibility that they caused livestock deaths.

In view of the many unknowns and unresolved problems relating blooms of toxic algae, it is impossible to suggest any recommendations insuring against the occurrence of toxic algae in livestock waters.

Recommendation

The use for livestock of waters bearing heavy growths of blue green algae should be avoided.

Radionuclides

Surface and groundwaters acquire radioactivity from natural sources, from fallout resulting from atmospheric nuclear detonations, from mining or processing uranium, or as the result of the use of isotopes in medicine, scientific research, or industry.

All radiation is regarded as harmful, and any unnecessary exposure to it should be avoided. Experimental work on the biological half-lives of radionuclides and their somatic and genetic effects on animals have been briefly reviewed by McKee and Wolf (1963).[193] Because the rate of decay of a radionuclide is a physical constant that cannot be changed, radioactive isotopes must be disposed of by dilution or by storage and natural decay. In view of the variability in half-lives of the many radioisotopes, the nature of their radioactive emissions, and the differences in metabolism of various elements by different animals, the results of animal experimentation do not lend themselves easily to the development of recommendations.

Based on the recommendations of the U. S. Federal Radiation Council (1960,[257] 1961[258]), the Environmental Protection Agency will set drinking water standards for radionuclides (1972),[145] to establish the intake of radioactivity from waters that when added to the amount from all other sources will not likely be harmful to man.

Recommendation

In view of the limited knowledge of the effect of radionuclides in water on domestic animals, it is recommended that the Federal Drinking Water Standards be used for farm animals as well as for man.

PESTICIDES (IN WATER FOR LIVESTOCK)

Pesticides include a large number of organic and inorganic compounds. The United States production of synthetic organic pesticides in 1970 was 1,060 million pounds consisting almost entirely of insecticides (501 million pounds), herbicides (391 million pounds), and fungicides (168 million pounds). Production data for inorganic pesticides was limited. Based on production, acreage treated, and use patterns, insecticides and herbicides comprise the major agricultural pesticides (Fowler 1972).[279] Of these, some can be detrimental to livestock. Some have low solubility in water, but all cause problems if accidental spillage produces high concentrations in water, or if they become adsorbed on colloidal particles subsequently dispersed in water.

Insecticides are subdivided into three major classes of compounds including methylcarbamates, organophosphates and chlorinated hydrocarbons. Many of these substances produce no serious pollution hazards, because they are nonpersistent. Others, such as the chlorinated hydrocarbons, are quite persistent in the environment and are the pesticides most frequently encountered in water.

Entry of Pesticides into Water

Pesticides enter water from soil runoff, direct application, drift, rainfall, spills, or faulty waste disposal techniques. Movement by erosion of soil particles with adsorbed pesticides is one of the principal means of entry into water. The amount carried in runoff water is influenced by rates of application, soil type, vegetation, topography, and other factors. Because of strong binding of some pesticides on soil particles, water pollution by pesticides is thought to occur largely through the transport of chemicals adsorbed to soil particles (Lichtenstein et al. 1966).[281] This mechanism may not always be a major route. Bradley et al. (1972)[269] observed that when 13.4 kg/hectare DDT and 26.8 kg/hectare toxophene were applied to cotton fields, only 1.3 and 0.61 per cent, respectively, of the amounts applied were detected in natural runoff water over an 8-month period.

Pesticides can also enter the aquatic environment by direct application to surface waters. Generally, this use is to control mosquito larvae, nuisance aquatic weeds, and, as in several southern states, to control selected aquatic fauna such as snails (Chesters and Konrad 1971).[271] Both of these pathways generally result in contamination of surface waters rather than groundwaters.

Precipitation, accidental spills, and faulty waste disposal are less important entry routes. Pesticides detected in rainwater include DDT, DDD, DDE, dieldrin, alpha-BHC and gamma-BHC in extremely minute concentrations (i.e., in the order of 10^{-12} parts or the nanograms per liter level) (Weibel et al. 1966,[295] Cohen and Pinkerton 1966,[274] Tarrant and Tatton 1968[291]). Spills and faulty waste disposal techniques are usually responsible for short-term, high-level contamination.

The amount of pesticide actually in solution, however, is governed by a number of factors, the most important probably being the solubility of the molecule. Chlorinated hydrocarbon insecticides, for example, have low solubility in water (Freshwater Appendix II-D). Cationic pesticides (i.e., paraquat and diquat) are rapidly and tightly bound to soil particles and are inactivated (Weed Society of America 1970).[291] Most arsenical pesticides form insoluble salts and are inactivated (Woolson et al. 1971).[297] A survey of the water and soil layers in farm ponds indicates higher concentrates of pesticides are associated with the soil layers that interface with water than in the water *per se*. In an extensive survey of farm water sources (U. S. Dept. of Agriculture, Agricultural Research Service 1969a,[292] hereafter referred to as Agriculture Research Service 1969a[267]), analysis of sediment showed residues in the magnitude of decimal fractions of a microgram per gram (µg g) to a high of 4.90 µg g DDT and its DDE and DDD degradation compounds. These were the principal pesticides found in sediment. Dieldrin and endrin were also detected in sediment in two study areas where surface drainage water entered farm ponds from an adjacent field.

Pesticides Occurrence in Water

Chlorinated hydrocarbon insecticides are the pesticides most frequently encountered in water. They include DDT and its degradation products DDE and DDD, dieldrin, endrin, chlordane, aldrin, and lindane. In a pesticide monitoring program conducted from 1957 to 1965, Breidenbach et al. (1967)[270] concluded that dieldrin was present in all sampled river basins at levels from 1 to 22 nanograms (ng) liter. DDT and its metabolites were found to occur in most surface waters, while levels of endrin in the lower

Mississippi decreased from a high of 214 ng/l in 1963 to a range of 15 to 116 ng/l in 1965. Results of monitoring studies conducted by the U. S. Department of Agriculture (Agricultural Research Service 1969a)[267] from 1965 to 1967 indicated that only very small amounts of pesticides were present in any of the sources sampled. The most prevalent pesticides in water were DDT, its metabolites DDD and DDE, and dieldrin. Levels detected were usually below one part per billion. The DDT family, dieldrin, endrin, chlordane, lindane, heptachlor epoxide, trifluralin, and 2,4-D, were detected in the range of 0.1 to 0.01 μg/l. In a major survey of surface waters in the United States conducted from 1965 to 1968 for chlorinated hydrocarbon pesticides (Lichtenberg et al. 1969),[282] dieldrin and DDT (including DDE and DDD) were the compounds most frequently detected throughout the 5-year period. After reaching a peak in 1966, the total number of occurrences of all chlorinated hydrocarbon pesticides decreased sharply in 1967 and 1968.

A list of pesticides most likely to occur in the environment and, consequently, recommended for inclusion in monitoring studies, was developed by the former Federal Committee on Pesticide Control (now Working Group on Pesticides). This list was revised (Schechter 1971)[290] and expanded to include those compounds (1) whose persistence is of relatively long-term duration; (2) whose use patterns is large scale in terms of acreage; or (3) whose inherent toxicity is hazardous enough to merit close surveillance. The primary list includes 32 pesticides or classes of pesticides (i.e. arsenical pesticides, mercurial pesticides, and several dithiocarbamate fungicides) recommended to be monitored in water. A secondary list of 17 compounds was developed for consideration, if monitoring activities are expanded in the future. The pesticides found on the primary list would be those most likely to be encountered in farm water supplies (see Freshwater Appendix II-D).

Toxicological Effects of Pesticides on Livestock

Mammals generally have a greater tolerance to pesticides than birds and fish. However, the increased use of pesticides in agriculture, particularly the insecticides, presents a potential hazard to livestock. Some compounds such as the organophosphorous insecticides can be extremely dangerous, especially when mishandled or wrongly used. To date, however, there actually have been very few verified cases of livestock poisoning from pesticides (Papworth 1967).[287] In the few instances reported, the cause of livestock poisoning usually has been attributed to human negligence. For livestock, pesticide classes that may pose possible hazards are the acaricides, fungicides, herbicides, insecticides, molluscides, and rodenticides (Papworth 1967).[287]

Acaricides intended for use on crops and trees usually have low toxicity to livestock. Some, such as technical chlorobenzilate, have significant toxicity for mammals. The acute oral LD50 in rats is 0.7 g/kg of body weight (Papworth 1967).[287] With fungicides, the main hazard to livestock apparently is not from the water route, but from their use as seed dressings for grain. Of the types used, the organomercury compounds and thiram are potentially the most dangerous (McEntee 1950,[283] Weibel et al. 1966[295]). The use of all organomercury fungicides is restricted by the Environmental Protection Agency (Office of Pesticides, Pesticides Regulation Division 1972).[277] Consequently, the possible hazard to livestock from these compounds has, for most purposes, been eliminated.

Of the herbicides in current use, the dinitro compounds pose the greatest hazard to livestock. Dinitroorthocresol (DNC or DNOC) is probably the most used member of this group. In ruminants, however, DNC is destroyed rapidly by the rumen organisms (Papworth 1967).[287] These compounds are very persistent, up to two years, and for livestock the greatest hazard is from spillages, contamination of vegetation, or water. In contrast, the phenoxyacetic acid derivatives (2,4-D, MCPA) are comparatively harmless. Fertig (1953)[278] states that suspected poisoning of livestock or wildlife by phenoxy herbicides could not be substantiated in all cases carefully surveyed. The hazards to livestock from hormone weed killers are discussed by Rowe and Hymas (1955),[289] and dinitrocompounds by McGirr and Papworth (1953)[284] and Edson (1954).[276] The possible hazards from other herbicides are reviewed by Papworth (1967)[287] and Radeleff (1970).[288]

Of the classes of insecticides in use, some pose a potential hazard to livestock, while others do not. Insecticides of vegetable origin such as pyrethrins and rotenones, are practically non-toxic to livestock. Most chlorinated hydrocarbons are not highly toxic to livestock, and none is known to accumulate in vital organs. DDT, DDD, dilan, methoxychlor, and perthane are not highly toxic to mammals, but some other chlorinated hydrocarbons are quite toxic (Papworth 1967,[287] Radeleff 1970[288]). The insecticides that are potentially the most hazardous are the organophosphorus compounds causing chlorinesterase inhibition. Some, such as mipafax, induce pathological changes not directly related to cholinesterase inhibition (Barnes and Denz 1953).[268] Liquid organophosphorus insecticides are absorbed by all routes, and the lethal dose for most of these compounds is low (Papworth 1967,[287] Radeleff 1970[288]).

Pesticides in Drinking Water for Livestock

The subgroup on contamination in the Report of the Secretary's Commission on Pesticides and Their Relationship to Environmental Health (U.S. Dept. of Health, Education, and Welfare 1969)[294] examined the present knowledge on mechanisms for dissemination of pesticides in the environment, including the water route. There have been no reported cases of livestock toxicity resulting from pesticides in water. However, they conclude that the possibility of contamination and toxicity from pesticides is real because of indiscriminate, uncontrolled and excessive use.

Pesticide residues in farm water supplies for livestock and related enterprises are undesirable and must be reduced or eliminated whenever possible. The primary problem of reducing levels of pesticides in water is to locate the source of contamination. Once located, appropriate steps should be taken to eliminate the source.

Some of the properties and concentrations of pesticides found in water are shown in Table V-4. Although many pesticides are readily broken down and eliminated by livestock with no subsequent toxicological effect, the inherent problems associated with pesticide use include the accumulation and secretion of either the parent compound or its degradation products in edible tissues and milk (Kutches et al. 1970).[280] Consequently, pesticides consumed by livestock through drinking water may result in residues in fat and certain produce (milk, eggs, wool), depending on the level of exposure and the nature of the pesticide. There is also a possibility of interactions between insecticides and drugs, especially in animal feeds (Conney and Hitchings 1969).[275]

Nonpolar lipophilic pesticides such as the chlorinated hydrocarbon insecticides (DDT, lindane, endrin, and others) tend to accumulate in fatty tissue and may result in measurable residues. Polar, water soluble pesticides and their metabolic derivatives are generally excreted in the urine soon after ingestion. Examples of this class would include most of the phosphate insecticides and the acid herbicides (2,4-D; 2,4,5-T; and others). Approximately 96 per cent of a dose of 2,4-D fed to sheep was excreted unchanged in the urine and 1.4 per cent in the feces in 72 hours (Clark et al. 1964).[273] Feeding studies (Claborn et al. 1960)[272] have shown that when insecticides were fed to beef cattle and sheep as a contaminant in their feed at dosages that occur as residues on forage crops, all except methoxychlor were stored in the fat. The levels of these insecticides in fat decreased after the insecticides were removed from the animals' diets. When poultry were exposed to pesticides either by ingestion of contaminated food or through the use of pesticides in poultry houses, Whitehead (1971)[296] observed that the toxicities to birds of the substances used varied greatly. However, nonlethal doses may affect growth rate, feed conversion efficiency, egg production, egg size, shell thickness, and viability of the young. Although the effects of large doses may be considerable, Whitehead concluded that little is known about the impairment of production at low rates commonly used in agricultural practice.

TABLE V-4—Some Properties, Criteria, and Concentrations of Pesticides Found in Water

	Solubility µg/liter	Toxicity LD50 mg/kg	Maximum concentration[a] µg/l
aldrin		38	0.085
dieldrin	110	46	0.407
endrin	160	10	0.133
heptachlor	56	130	0.048
heptachlor epoxide	350		0.067
DDT	1.2	113	0.316
DDE			0.050
DDD			0.840
2,4-D[b]	60,000	300-1000	

[a] Maximum concentration of pesticide found in surface waters in the United States, from Lichtenberg et al. (1969)[283].

[b] Refers to the herbicide family 2,4-D; 2,4,5-T; and 2,4,5-TP.

Elimination of fat soluble pesticides from contaminated animals is slow. Urinary excretion is insignificant and elimination in feces is slow. The primary route of excretion in a lactating animal is through milk. The lowest concentrations of pesticides in feeds that lead to detectable residues in animal tissues or products exceed the amounts found in water by a factor of 10,000. However, at the comparatively high dosage rates given in feeds, certain trends are apparent. Cows fed DDT in their diet at rates of 0.5, 1.0, 2.0, 3.0, and 5.0 mg/kg exhibited residues in milk at all feeding levels except at 0.5 mg/kg. As the DDT feed levels increased, contamination increased (Zweig et al. 1961).[298] When cows were removed from contaminated feeds, the amount of time required for several pesticides to reach the non-detectable level was recorded (Moubry et al. 1968).[286] Dieldrin had the longest retention time in milk, approximately 100 days. DDT and its analogs, BHC, lindane, endrin, and methoxychlor followed in that order. It should be emphasized that levels found in farm water supplies do not make a significant contribution to animal body burdens of pesticides compared to amounts accumulated in feeds.

Table V-4 shows the toxicity of some important pesticides. Assuming the average concentration of any pesticide in water is 0.1 µg/l, and the average daily consumption of water by dairy or beef cattle is 60 liters per day, then the average daily intake of DDT would be 0.006 mg. Further, assuming that the average body weight for dairy or beef cattle is 450 kg and the LD50 for DDT is 113 mg/kg (Table V-4), then 50 grams would have to be consumed to approach the dose that would be lethal to 50 per cent of the animals. If a steer were maintained on this water for 1,000 days, then it would have ingested about 1/10,000 of the reported LD50. For endrin (LD50 = 10 mg/kg), cattle would ingest 1/1,000 of the established LD50. The safety margin is probably greater than indicated, because the calculations assume that all of the insecticide is retained unaltered during the total ingestion period. DDT is known to be degraded to a limited extent by bovine rumen fluid and by rumen microorganisms. For sheep, swine, horses, and poultry, the average daily water intake in liters is about 5, 10, 40, and 0.2, respectively. Consequently, their intake would be substantially less.

Fish as Indicators of Water Safety

The presence of fish may be an excellent monitor for toxic levels of pesticides in livestock water supplies. There are numerous and well documented examples in the literature of the biological magnification of persistent pesticides

TABLE V-5—*Examples of Fish as Indicators of Water Safety for Livestock*

Material	Toxic-levels mg/l for fish	Toxic effects on animals
Aldrin	0.02	3 mg/kg food (poultry).
Chlordane	1.0 (sunfish)	91 mg/kg body weight in food (cattle).
Dieldrin	0.025 (trout)	25 mg/kg food (rats).
Dipterex	50.0	10.0 mg/kg body weight in food (calves).
Endrin	0.003 (bass)	3.5 mg/kg body weight in food (chicks).
Ferban, fermate	1.0 to 4.0	
Methoxychlor	0.2 (bass)	14 mg/kg alfalfa hay, not toxic (cattle).
Parathion	2.0 (goldfish)	75 mg/kg body weight in food (cattle).
Pentachlorophenol	0.35 (bluegill)	60 mg/l drinking water not toxic (cattle).
Pyrethrum (allethrin)	2.0 to 10.0	1,400 to 2,800 mg/kg body weight in food (rats).
Silvex	5.0	500 to 2,000 mg/kg body weight in food (chicks).
Toxaphene	0.1 (bass)	35 to 110 mg/kg body weight in food (cattle).

McKee and Wolf, 1963[295].

by fish and other aquatic organisms (See Sections III and IV on Freshwater and Marine Aquatic Life and Wildlife.) Because of the lower tolerance levels of these aquatic organisms for persistent pesticides such as chlorinated hydrocarbon insecticides, mercurial compounds, and heavy metal fungicides, the presence of living fish in agricultural water supplies would indicate their safety for livestock (McKee and Wolf 1963).[295] Some examples of individual effects of pesticides upon fish compared to animal species are shown in Table V-5. These data indicate that fish generally have much lower tolerance for commonly used pesticides than do livestock and poultry.

Recommendation

Feeding studies indicate no deleterious effects of reported pesticide residues in livestock drinking water on animal health. To prevent unacceptable residues in animal products, the maximum levels proposed in the pesticide section of the Panel of Public Water Supplies are recommended for farm animal water supplies.

PATHOGFNS AND PARASITIC ORGANISMS

Microbial Pathogens

One of the most significant factors in the spread of infectious diseases of domesticated animals is the quality of water which they consume. In many instances the only water available to livestock is from surface sources such as ponds, waterholes, lakes, rivers and creeks. Not infrequently these sources are contaminated by animals which wade to drink or stand in them seeking refuge from pests. Contamination with potential disease-producing organisms comes from surface drainage originating in corrals, feed lots, or pastures in which either sick or carrier animals are kept.

Direct evidence relating the occurence of animal pathogens in surface waters and disease outbreaks is limited. However, water may be a source for listeriosis caused by *Listeria monocytogenes* (Larsen 1964)[302] and erysipelas caused by *Erysipelothrix rhusiopathiae* (Wood and Packer in press 1972).[310] Tularemia of animals is not normally waterborne, but the organism *Pasteurella tularensis* has been isolated from waters in the United States (Parker et al. 1951,[303] Seghetti 1952).[305] Enteric microorganisms, including the vibrios (Wilson and Miles 1966)[309] and amoebae, have a long record as water polluting agents.

The *Escherichia-Enterobacter-Klebscilla* group of enterics are widely distributed in feed, water, and the general environment (Breed et al. 1957).[299] They sometimes cause urinary disease, abscesses, and mastitis in livestock. *Salmonella* are very invasive and the carrier state is easily produced and persistent, often without any general evidence of disease. Spread of the enterics outside the yards, pens, or pastures of infected livestock is a possibility, but the epidemiology and ecology of this problem are not clear.

In the United States, leptospirosis is probably the most intimately water-related disease problem (Gillespie et al. 1957,[301] Crawford et al. 1969[300]). The pathogenic leptospira leave the infected host through urine and lack protection against drying. Direct animal-to-animal spread can occur through urine splashed to the eyes and nostrils of another animal.

Infection by leptospirosis from water often is direct; that is, contaminated water infects animals that consume it or come into contact with it.

Van Thiel (1948)[308] and Gillespie et al. (1957)[301] pointed out that mineral composition and pH of water are factors affecting continued mobility of voided leptospira. Most episodes of leptospirosis can be traced to ponds, ricefields, and natural waters of suitable pH and mineral composition. For leptospira control, livestock must not be allowed to wade in contaminated water. Indirect contamination of water through sewage is unlikely, although free-living leptospira may occur in such an environment.

The Genus *Clostridium* is comprised of many species (Breed et al. 1957),[299] some of which have no pathogenic characteristics. Some such as *Clostridium perfrigens* and *Cl. tetani* may become adapted to an enteric existence in animals. Almost all of them are soil adapted. Water has a vital role in environments favorable for anaerobic infections caused by *Clostridia*.

Management of water to avoid oxygen depletion serves to control the anaerobic problem. Temporary or permanent areas of anaerobic water environment are dangerous to livestock. Domestic animals should be prevented from consuming water not adequately oxygenated.

One of the best examples of water-related disease is bacillary hemoglobinuria, caused by an organism *Cl. hemolyticum* found in western areas of North and South America. It has been linked with liver fluke injury, but is not dependent on the presence of flukes. Of particular concern has been the spread of this disease to new areas in the western states. As described by Van Ness and Erickson (1964),[307] each new premise is an endemic area which has an alkaline, anaerobic soil-water environment suitable for the organism. This disease has made its appearance in new areas of the West when these areas are cleared of brush and irrigated. To avoid this problem, western irrigation waters should be managed to avoid cattail marshes, hummock grasses, and other environments of prolonged saturation.

Anthrax in livestock is a disease of considerable concern. The organism causing anthrax, *Bacillus anthracis*, may occur in soils with pH above 6.0. The organism forms spores which, in the presence of adequate soil nutrients, vegetate and grow. The spread of disease by drinking water containing spores has never been proved. Bits of hide and hair waste may be floated by water downstream from manufacturing plants, but very few outbreaks have been reported from these sources. The disease is associated with the water from pastures where the grass has been killed (Van Ness 1971).[306] The killed grass is brown rather than blackened, a significant difference from water drowned vegetation in general.

The epidemiology of virus infections tends to incriminate direct contact: e.g., fomites, mechanical, and biological vectors, but seldom water supplies. Water used to wash away manure prior to the use of disinfectants or other biological control procedure may carry viruses to the general environment.

Viruses are classified by size, type of nucleic acid, structure, ether sensitivity, tissue effects (which includes viruses long known to cause recognizable diseases, such as pox and hog cholera), and by other criteria. Only the ether-resistant viruses, such as those causing polio and foot and mouth disease in cattle, appear to present problems in natural water (Prier 1966).[304]

Parasitic Organisms

Parasitic protozoa include numerous forms which are capable of causing serious livestock losses. Most outbreaks follow direct spread among animals. Water contaminated with these organisms or their cysts becomes an indirect factor in spread of infection.

Some of the most important parasitic forms are the various flukes which develop as adult forms in man and livestock. Important ecological factors include presence of snails and vegetation in the water, or vegetation covered by intermittent overflow. This problem is very serious in irrigated areas, but only when snails or other intermediate hosts are available for the complete life cycle. Fluke eggs passed by the host, usually in the manure (some species, in the urine), enter the water and hatch into *miracidia*. These seek out a snail or other invertebrate host where they develop into *sporocysts*. These transform into redia which in turn may form other redia or several *cercariae*. The cercariae leave the snail and swim about the water where they may find the final host, or may encyst on vegetation to be eaten later. The life cycle is completed by maturing in a suitable host and establishment of an exit for eggs from the site of the attachment.

Roundworms include numerous species which may use water pathways in their life cycle. Free-living nematodes can sometimes be found in a piped water supply, but are probably of little health significance. Moisture is an important factor in the life cycle of many parasitic roundworms and livestock are maintained in an environment where contamination of water supplies frequently occurs. It is usually thought that roundworm eggs are eaten but water-saturated environments provide ideal conditions for maintaining populations of these organisms and their eggs.

Parasitic roundworms probably evolved through evolutionary cycles exemplified by the behavior of the genus *Strongyloides*. *Strongyloides* spread along drainageways through the washdown of concrete feeding platforms and other housing facilities for livestock.

The Guinea worm, *Dracunculus*, is dependent upon water, because the adult lays eggs only when the host comes in contact with water. Man, dogs, cats, or various wild mammals may harbor the adult, and the larvae develop in *Cyclops*. The life cycle is thus maintained in a water environment when the *Cyclops* is swallowed by another suitable host.

Eggs of "horsehair worms" are laid by the adult in water or moist soil. The larvae encyst and if eaten by an appropriate insect will continue development to the adult stage. Worms do not leave the insect unless they can enter water.

The prevention of water-borne diseases and parasitisms in domestic animals depends on interruption of the organisms' life cycle. The most effective means is to keep livestock out of contaminated water. Treatment for the removal of the pathogen or parasite from the host and destruction of the intermediate host are measures of control.

WATER FOR IRRIGATION

Irrigation farming increases productivity of croplands and provides flexibility in alternating crops to meet market demands. Early irrigation developments in the arid and semiarid West were largely along streams where only a small part of the total annual flow was put to use. Such streams contained dissolved solids accumulated through the normal leaching and weathering processes with only slight additions or increases in concentrations resulting from man's activities. Additional uses of water resources have in many cases concentrated the existing dissolved solids, added new salts, contributed toxic elements, microbiologically polluted the streams, or in some other way degraded the quality of the water for irrigation. Water quality criteria for irrigation has become increasingly significant as new developments in water resources occur.

Soil, plant, and climate variables and interactions must be considered in developing criteria for evaluation of irrigation water quality. A wide range of suitable water characteristics is possible even when only a few variables are considered. These variables are important in determining the quality of water that can be used for irrigation under specific conditions.

The physicochemical properties of a soil determine the root environment that a plant encounters following irrigation. The soil consists of an organo-mineral complex that has the ability to react both physically and chemically with constituents present in irrigation water. The degree to which these added constituents will leach out of a soil, remain available to plants in the soil, or become fixed and unavailable to plants, depends largely on the soil characteristics.

Evapotranspiration by plants removes water from the soil leaving the salts behind. Since uptake by plants is negligible, salts accumulate in the soil in arid and semiarid areas. A favorable salt balance in the root zone can be maintained by leaching, through the use of irrigation water in excess of plant needs. Good drainage is essential to prevent a rising water table and salt accumulation in the soil surface and to maintain adequate soil aeration.

In irrigated areas, a water frequently exists at some depth below the ground surface, with an unsaturated condition existing above it. During and immediately following periods of precipitation or irrigation, water moves downward through the soil to the water table. At other times, water is lost through evaporation from the soil surface, and transpiration from plants (evapotranspiration) may reverse the direction of flow in the soil, so that water moves upward from the water table by capillary flow. The rate of movement is dependent upon water content, soil texture, and structure. In humid and subhumid regions, this capillary rise of water in the soil is a valuable water source for use by crops during periods of drought.

Even under favorable conditions of soil, drainage, and environmental factors, too sparing applications of high quality water with total dissolved solids of less than 100 mg/l would ultimately damage sensitive crops such as citrus fruit; whereas with adequate leaching, waters containing 500 to 1,000 mg/l might be used safely. Under the same conditions, certain salt-tolerant field crops might produce economic returns using water with more than 4,000 mg/l. Criteria for judging water quality must take these factors into account.

The need for irrigation for optimum plant growth is determined also by rainfall and snow distribution; and by temperature, radiation, and humidity. Irrigation must be used for intensive crop production in arid and semiarid areas and must supplement rainfall in humid areas. (See Specific Irrigation Water Considerations below.)

The effects of water quality characteristics on soils and on plant growth are directly related to the frequency and amount of irrigation water applied. The rate at which water is lost from soils through evapotranspiration is a direct function of temperature, solar radiation, wind, and humidity. Soil and plant characteristics also influence this water loss. Aside from water loss considerations, water stress in a plant, as affected by the rate of evapotranspiration, will determine the plant's reaction to a given soil condition. For example, in a saline soil at a given water content, a plant will usually suffer more in a hot, dry climate than in a cool, humid one. Considering the wide variation in the climatic and soil variables over the United States, it is apparent that water quality requirements also vary considerably.

Successful sustained irrigated agriculture, whether in arid regions or in subhumid regions, or other areas, requires skillful water application based upon the characteristics of the land, water, and the requirements of the crop. Through proper timing and adjustment of frequency and volumes of water applied, detrimental effects of poor quality water may often be mitigated.

WATER QUALITY CONSIDERATIONS FOR IRRIGATION

Effects on Plant Growth

Plants may be adversely affected directly by either the development of high osmotic conditions in the plant substrate or by the presence of a phytotoxic constituent in the water. In general, plants are more susceptible to injury from dissolved constituents during germination and early growth than at maturity (Bernstein and Hayward 1958).[315] Plants affected during early growth may result in complete crop failure or severe yield reductions. Effects of undesirable constituents may be manifested in suppressed vegetative growth, reduced fruit development, impaired quality of the marketable product, or a combination of these factors. The presence of sediment, pesticides, or pathogenic organisms in irrigation water, which may not specifically affect plant growth, can affect the acceptability of the product. Another aspect to be considered is the presence of elements in irrigation water that are not detrimental to crop production but may accumulate in crops to levels that may be harmful to animals or humans.

Where sprinkler irrigation is used, foliar absorption or adsorption of constituents in the water may be detrimental to plant growth or to the consumption of affected plants by man or animals. Where surface or sprinkler irrigation is practiced, the effect of a given water quality on plant growth is determined by the composition of the soil solution. This is the growth medium available to roots after soil and water have reacted.

Plant growth may be affected indirectly through the influence of water quality on soil. For example, the absorption by the soil of sodium from water will result in a dispersion of the clay fraction. The degree of dispersion will depend on the clay minerals present. This decreases soil permeability and often results in a surface crust formation that deters seed germination and emergence. Soils irrigated with highly saline water will tend to be flocculated and have relatively high infiltration rates (Bower and Wilcox 1965).[316] A change to waters of sufficiently lower salt content reduces soil permeability and rates of infiltration by dispersion of the clay fraction in the soil. This hazard increases when combined with high sodium content in the water. Much depends upon whether a given irrigation water is used continuously or occasionally.

Crop Tolerance to Salinity

The effect of salinity, or total dissolved solids, on the osmotic pressure of the soil solution is one of the most important water quality considerations. This relates to the availability of water for plant consumption. Plants have been observed to wilt in fields apparently having adequate water content. This is usually the result of high soil salinity creating a physiological drought condition. Specifically, the ability of a plant to extract water from a soil is determined by the following relationship:

$$\mathrm{TSS} = \mathrm{MS} + \mathrm{SS}$$

In this equation, (U.S. Department of Agriculture, Salinity Laboratory Staff 1954[337] hereafter referred to as Salinity Laboratory 1954[335]) the total soil suction (TSS) represents the force with which water in the soil is withheld from plant uptake. In simplified form, this factor is the sum of the matric suction (MS) or the physical attraction of soil for water, and the solute suction (SS) or the osmotic pressure of the soil water.

As the water content of the soil decreases due to evapotranspiration, the water film surrounding the soil particles becomes thinner and the remaining water is held with increasingly greater force (MS). Since only pure water is lost to the atmosphere during evapotranspiration, the salt concentration of soil solution increases rapidly during the drying process. Since the matric suction of a soil increases exponentially on drying, the combined effects of these two factors can produce critical conditions with regard to soil water availability.

In assessing the problem of plant growth, the salinity level of the soil solution must be evaluated. It is difficult to extract the soil solution from a moist soil within the range of water content available to plants. It has been demonstrated, however, that salinity levels of the soil solution and their resultant effects upon plant growth may be correlated with salinity levels of soil moisture at saturation. The quantity of water held in the soil between field capacity and the wilting point varies considerably from relatively low values for sandy soils to high values for soils high in clay content.

The U.S. Salinity Laboratory Staff (1954)[335] developed the technique of using a saturation extract to meet this need. Demineralized water is added to a soil sample to a point at which the soil paste glistens as it reflects light and flows slightly when the container is tipped. The amount of water added is reasonably related to the soil texture. For many soils, the water content of the soil paste is roughly twice that of the soil at field capacity and four times that at the wilting point. This water content is called the saturation percentage. When the saturated paste is filtered, the resultant solution is referred to as the saturation extract. The salt content of the saturation extract does not give an exact indication of salinity in the soil solution under field conditions, because soil structure has been destroyed; nor does it give a true picture of salinity gradients within the soil resulting from water extraction by roots. Although not truly depicting salinity in the immediate root environment, it does give a usable parameter that represents a soil salinity value that can be correlated with plant growth.

Salinity is most readily measured by determining the electrical conductivity (EC) of a solution. This method relates to the ability of salts in solution to conduct electricity and results are expressed as millimhos ($\text{mhos} \times 10^{-3}$) per centimeter (cm) at 25 C. Salinity of irrigation water is expressed in terms of EC, and soil salinity is indicated by the electrical conductivity of the saturation extract (EC_e). See Table V-6.

Temperature and wind effects are especially important as they directly affect evapotranspiration. Periods of high temperature or other factors such as dry winds, which increase evapotranspiration rates, not only tend to increase soil salinity but also create a greater water stress in the plant. The effect of climate conditions on plant response to salinity was demonstrated by Magistad and his associates (1943).[321] Some of these effects can be alleviated by more frequent irrigation to maintain safer levels of soil salinity.

Plants vary in their tolerance to soil salinity, and there are many ways in which salt tolerance can be appraised. Hayward and Bernstein (1958)[321] point out three: (1) Test the ability of a plant to survive on saline soils. Salt tolerance based primarily on this criterion of survival has limited application in irrigation agriculture but is a method of appraisal that has been used widely by ecologists. (2) Test the absolute yield of a plant on a saline soil. This criterion has the greatest agronomic significance. (3) Relate the yield on saline soil to nonsaline soil. This criterion is useful for comparing dissimilar crops whose absolute yields cannot be compared directly.

The U. S. Salinity Laboratory Staff (1954)[335] has used the third criterion in establishing the list of salt tolerance of various crops shown in Table V-6. These salt tolerance values are based upon the conductivity of the saturation extract (EC_e) expressed in mmhos/cm at which a 50 per cent decrement in yield may be expected when compared to

TABLE V-6—Relative Tolerance of Crop Plants to Salt, (Listed in Decreasing Order of Tolerance[a])

High salt tolerance	Medium salt tolerance	Low salt tolerance
	VEGETABLE CROPS	
$EC_e \times 10^3 = 12$	$EC_e \times 10^3 = 10$	$EC_e \times 10^3 = 4$
Garden beets	Tomato	Radish
Kale	Broccoli	Celery
Asparagus	Cabbage	Green beans
Spinach	Bell pepper	
	Cauliflower	
	Lettuce	
	Sweet corn	
	Potatoes (White Rose)	
	Carrot	
	Onion	
	Peas	
	Squash	
	Cucumber	
$EC_e \times 10^3 = 10$	$EC_e \times 10^3 = 4$	$EC_e \times 10^3 = 3$
	FIELD CROPS	
$EC_e \times 10^3 = 16$	$EC_e \times 10^3 = 10$	$EC \times 10^3 = 4$
Barley (grain)	Rye (grain)	Field beans
Sugar beet	Wheat (grain)	
Rape	Oats (grain)	
Cotton	Rice	
	Sorghum (grain)	
	Corn (field)	
	Flax	
	Sunflower	
	Castorbeans	
$EC \times 10^3 = 10$	$EC_e \times 10^3 = 6$	
	FRUIT CROPS	
Date palm	Pomegranate	Pear
	Fig	Apple
	Olive	Orange
	Grape	Grapefruit
	Cantaloupe	Prune
		Plum
		Almond
		Apricot
		Peach
		Strawberry
		Lemon
		Avocado
	FORAGE CROPS (in decreasing order tolerance)	
$EC_e \times 10^3 = 18$	$EC_e \times 10^3 = 12$	$EC_e \times 10^3 = 4$
Alkali sacaton	White sweet clover	White Dutch clover
Saltgrass	Yellow sweet clover	Meadow foxtail
Nuttall alkaligrass	Perennial ryegrass	Alsike clover
Bermuda grass	Mountain brome	Red clover
Rhodes grass	Strawberry clover	Ladino clover
Rescue grass	Dallis grass	Burnet
Canada wildrye	Sudan grass	
Western wheatgrass	Hubam clover	
Barley (hay)	Alfalfa (California common)	
Bridsfoot trefoil	Tall fescue	
	Rye (hay)	
	Wheat (hay)	
	Oats (hay)	
	Orchardgrass	
	Blue grama	
	Meadow fescue	
	Reed canary	
	Big trefoil	
	Smooth brome	
	Tall meadow oatgrass	
	Cicer milkvetch	
	Sourclover	
	Sickle milkvetch	
$EC_e \times 10^3 = 12$	$EC_e \times 10^3 = 4$	$EC_e \times 10^3 = 2$

[a] The numbers following $EC_e \times 10^3$ are the electrical conductivity values of the saturation extract in millimhos per centimeter at 25 C associated with 50-per cent decrease in yield.

Salinity Laboratory Staff 1954[335].

TABLE V-7—Soil Salinities in Root Zone at which Yield Reductions become Significant

Crop	Electrical conductivity of saturation extracts (EC_e) at which yields decrease by about 10 per cent[a]
	mmh/cm at 25 C
Date palm	8
Pomegranate Fig Olive	4-6[b]
Grape	4
Muskmelon	3.5
Orange, grapefruit, lemon[c]	3-2.5
Apple, pear	2.5
Plum, prune, peach, apricot, almond	2.5
Boysenberry, blackberry, raspberry[c]	2.5-1.5
Avocado	2
Strawberry	1.5

[a] In gypsiferous soils, EC_e readings for given soil salinities are about 2 mmh/cm higher than for nongypsiferous soils. Date palm would be affected at 10 mmh/cm, grapes at 6 mmh/cm, etc. on gypsiferous soils.

[b] Estimated.

[c] Lemon is more sensitive than orange and grapefruit; raspberry more than boysenberry and blackberry. Bernstein 1965b[314].

TABLE V–8—Salt Tolerance of Ornamental Shrubs

(Maximum EC_e's tolerated)

Tolerant	Moderately tolerant	Sensitive	Very sensitive
6–10	4–6	2–4	2
Carissa grandiflora (Natal plum) Bougainvillea spectabilis (Bogainvillea) Nerium oleander (oleander) Rosmarinus lockwoodi (Rosmary) Dodonea viscosa atropurpurea Callistemon viminalis (bottlebrush)	Dracaena endivisa Thuja orientalis (arbor vitae) Juniperus chinensis (spreading juniper) Euonymus japonica grandiflora Lantana camara Elaeagnus pungens (silverberry) Xylosma senticosa Pittosporum tobira Pyracantha Graberi Ligustrum lucidum (Texas privet) Buxus microphylla japonica (Japanese boxwood)	Hibiscus rosa-sinensis var. Brilliante Nandina domestica (heavenly bamboo) Trachelospermum jasminoides (star jasmine) Viburnum tinus robustum	Ilex cornuta Burford (Burford holly) Hedera canariensis (Algerian ivy) Feijoa sellowiana (pineapple guava) Rosa sp. (var. Grenoble rose on Dr. Huey root)

Bernstein 1965b[314].

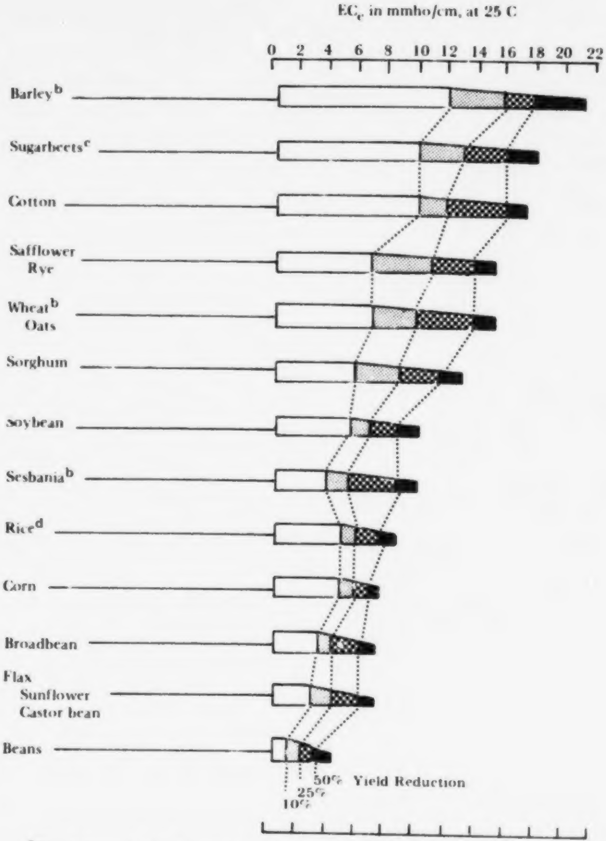
yields of that plant grown on a nonsaline soil under comparable growing conditions. Work has been done by many investigators, based upon both field and greenhouse research, to evaluate salt tolerance of a broad variety of plants. In general, where comparable criteria were used to assess salt tolerance, results obtained were most often in agreement. Recent work on the salt tolerance of fruit crops is shown in Table V-7, and for ornamentals in Table V-8.

Bernstein (1965a[313]) gave EC_e values causing 10, 25, and 50 per cent yield decrements for a variety of field and forage crops from late seeding stage to maturity, assuming that sodium or chloride toxicity was not a growth deterrent. These values are shown in Figures V-1, V-2, and V-3. The data suggested that the effects of EC_e values producing 10 to 50 per cent decrements (within a range of EC_e values of 8 to 10 mmh/cm for many crops) may be considered approximately linear, but for nearly all crops the rate of change

$$EC_e \frac{\Delta y}{\Delta EC_e},$$ either steepens or flattens slightly as the yield

decrements increase from less than 25 to more than 25 per cent. Bernstein (1965a)[313] also pointed out that most fruit crops were more sensitive to salinity than were field, forage, or vegetable crops. The data also illustrated the highly variable effect of EC_e values upon different crops and the nonlinear response of some crops to increasing concentrations of salt.

In considering salt tolerances of crops, EC_e values were used. These values were correlated with yields at field moisture content. If soils were allowed to dry out excessively between irrigations, yield reductions were much greater, since the total soil water stress is a function of both matric suction and solute suction and increases exponentially on drying (Bernstein 1965a).[313] Good irrigation management can minimize this hazard.



[a] The indicated salt tolerances apply to the period of rapid plant growth and maturation, from the late seeding stage upward. Crops in each category are ranked in order of decreasing salt tolerance. Width of the bar next to each crop indicates the effect of increasing salinity on yield. Crosslines are placed at 10, 25, and 50 per cent yield reductions. Approximate rank in order of decreasing salt tolerance is indicated for additional crops for most of which complete data are lacking. (Bower *personal communication* 1972)238

[b] Less tolerant during seedling stage. Salinity at this stage should not exceed 4 or 5 mmho/cm, EC_e.

[c] Sensitive during germination. Salinity should not exceed 3 mmho/cm during germination.

[d] Less tolerant during flowering and seed-set as well as during the seedling stage. Salinity at sensitive stages should not exceed 4 mmho/cm, EC_e of soil water.

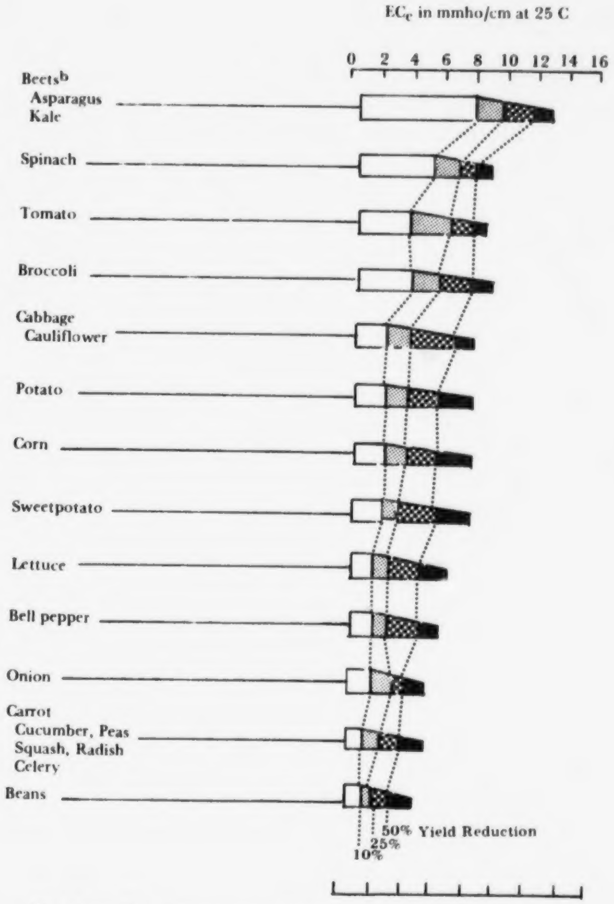
FIGURE V-1—Salt Tolerance of Field Crops[a]

Nutritional Effects

Plants require a blanced nutrient content in the soil solution to maintain optimum growth. Use of saline water for irrigation may or may not significantly upset this nutritional balance depending upon the composition, concentration, and volume of irrigation water applied.

Some of the possible nutritional effects were summarized by Bernstein (1965a)[313] as follows:

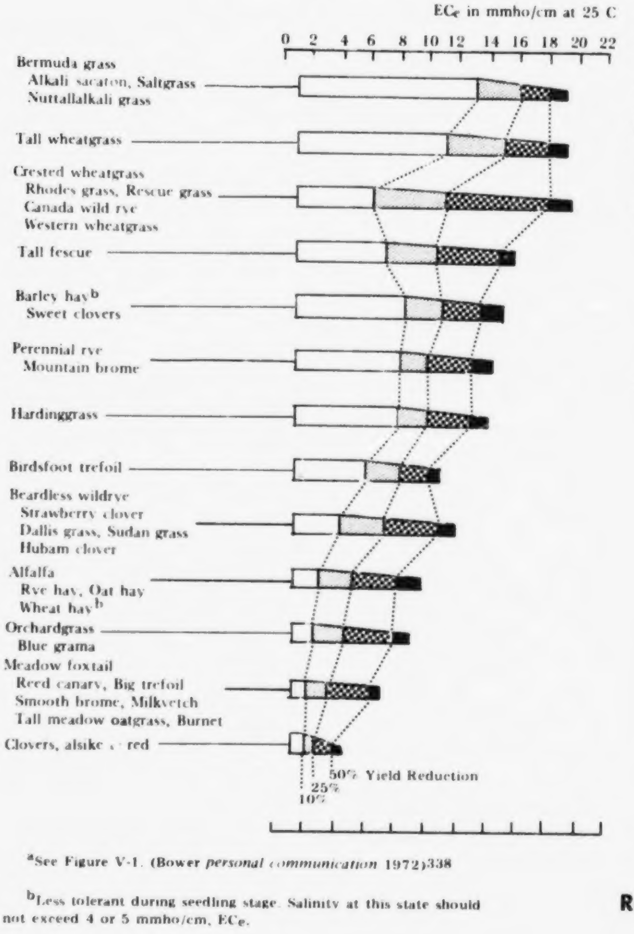
> High concentrations of calcium ions in the solution may prevent the plant from absorbing enough potassium, or high concentrations of other ions may affect the uptake of sufficient calcium.
>
> Different crops vary widely in their requirements for given nutrients and in their ability to absorb them. Nutritional effects of salinity, therefore, appear only in certain crops and only when a particular type of saline condition exists.
>
> Some varieties of a particular crop may be immune to nutritional disturbances, while other varieties are severely affected. High levels of soluble sulfate cause internal browning (a calcium deficiency symptom) in some lettuce varieties, but not in others. Similarly, high levels of calcium cause greater nutritional disturbances in some carrot varieties than in others. Chemical analysis of the plant is useful in diagnosing these effects.
>
> At a given level of salinity, growth and yield are depressed more when nutrition is disturbed than when nutrition is normal. Nutritional effects, fortunately, are not important in most crops under saline conditions; when they do occur, the use of better adapted varieties may be advisable.



[a]See Figure V-1. (Bower *personal communication* 1972)338

[b]Sensitive during germination. Salinity should not exceed 3 mmho/cm ECe during germination.

FIGURE V-3—Salt Tolerance of Vegetable Crops[a]



[a]See Figure V-1. (Bower *personal communication* 1972)338

[b]Less tolerant during seedling stage. Salinity at this state should not exceed 4 or 5 mmho/cm, ECe.

FIGURE V-2—Salt Tolerance of Forage Crops[a]

Recommendation

Crops vary considerably in their tolerance to soil salinity in the root zone, and the factors affecting

the soil solution and crop tolerance are varied and complex. Therefore, no recommendation can be given for these. For specific crops, however, it is recommended that the salt tolerance values (EC_e) for a saturation extract established by the U.S. Salinity Laboratory Staff be used as a guide for production.

Temperature

The temperature of irrigation water has a direct and indirect effect on plant growth. Each occurs when plant physiological functions are impaired by excessively high or excessively low temperatures. The exact water temperatures at which growth is severely restricted depends on method of water application, atmospheric conditions at the time of application, frequency of application, and plant species. All plant species have a tempeature range in which they develop best. These temperature limits vary with plant species.

Direct effect on plant growth from extreme temperature of the irrigation water occurs when the water is first applied. Plant damage results only from direct contact. Normally, few problems arise when excessively warm water is applied by sprinkler irrigation. The effect of the temperature of the water on the temperature of the soil is negligible. It has been demonstrated that warm water applied through a sprinkler system has attained ambient temperatures at the time it reaches the soil surface (Cline et al. 1969).[318] Water as warm as 130 F can be safely used in this manner. Cold water is harmful to plant growth when applied through a sprinkler system. It does not change in temperature nearly so much as the warm water. However, its effect is rarely lethal.

Surface applied water that is either very cold or very warm poses greater problems. Excessive warm water cannot be used for surface irrigation and cold water affects plant growth. The adverse effects of cold water on the growth of rice were suddenly brought to the attention of rice growers when cold water was first released from the Shasta Reservoir in California (Raney 1963).[332] Summer water temperatures were suddenly dropped from about 61 F to 45 F. Research is still proceeding, and some of the available information was recently reviewed by Raney and Mihara (1967).[334] Dams such as the Oroville Dam are now being planned so that water can be withdrawn from any reservoir depth to avoid the cold-water problem. Warming basins have been used (Raney 1959).[333] There are opportunities in planning to separate waters—the warm waters for recreation and agriculture, the cold waters for cold-water fish, salmon spawning, and other uses. The exact nature of the mechanisms by which damage occurs is not completely understood.

Indirect effect of the temperature of irrigation water on plant growth occurs as a result of its influence on the temperature of the soil. The latter affects the rate of water uptake, nutrient uptake, translocation of metabolites, and, indirectly, such factors as stomatal opening and plant water stress. All these phenomena are well documented. The effect of the temperature of the applied irrigation water on the temperature of the soil is not well described. This effect is probably quite small.

Conclusion

Present literature does not provide adequate data to establish specific temperature recommendations for irrigation waters. Therefore, no specific recommendations can be made at this time.

Chlorides

Chlorides in irrigation waters are not generally toxic to crops. Certain fruit crops are, however, sensitive to chlorides. Bernstein (1967)[312] indicated that maximum permissible chloride concentrations in the soil range from 10 to 50 milliequivalents (meq)/l for certain sensitive fruit crops (Table V-9). In terms of permissible chloride concentrations in irrigation water, values up to 20 meq/l can be used, depending upon environmental conditions, crops, and irrigation management practices.

Foliar absorption of chlorides can be of importance in sprinkler irrigation (Eaton and Harding 1959,[319] Ehlig and Bernstein 1959[320]). The adverse effects vary between evapo-

TABLE V-9—Salt Tolerance of Fruit Crop Varieties and Rootstocks and Tolerable Chloride Levels in the Saturation Extracts

Crop	Rootstock or variety		Tolerable levels of chloride in saturation extract
	Rootstocks		meq/l
Citrus	Rangpur lime, Cleopatra mandarin		25
	Rough lemon, tangelo, sour orange		15
	Sweet orange, citrange		10
Stone fruit	Marianna		25
	Lovell, Shalil		10
	Yunnan		7
Avocado	West Indian		8
	Mexican		5
	Varieties (V) and Rootstocks (R)		
Grape	Salt Creek, 1613-3	R	40
	Dog Ridge		30
	Thompson Seedless, Perlette	V	20
	Cardinal, Black Rose		10
	Varieties		
Berries	Boysenberry		10
	Olallie blackberry		10
	Indian Summer raspberry		5
Strawberry	Lassen		8
	Shasta		5

Bernstein 1967[312].

rative conditions of day and night and the amount of evaporation that can occur between successive wettings (i.e., time after each pass with a slowly revolving sprinkler). There is less effect with nighttime sprinkling and less effect with fixed sprinklers (applying water at a rapid rate). Concentrations as low as 3 meq/l of chloride in irrigation water have been found harmful when used on citrus, stone fruits, and almonds (Bernstein 1967).[312]

Conclusion

Permissible chloride concentrations depend upon type of crop, environmental conditions and management practices. A single value cannot be given, and no limits should be established, because detrimental effects from salinity per se ordinarily deter crop growth first.

Bicarbonates

High bicarbonate water may induce iron chlorisis by making iron unavailable to plants (Brown and Wadleigh 1955).[317] Problems have been noted with apples and pears (Pratt 1966)[330] and with some ornamentals (Lunt et al. 1956).[323] Although concentrations of 10 to 20 meq/l of bicarbonate can cause chlorosis in some plants, it is of little concern in the field where precipitation of calcium carbonate minimizes this hazard.

Conclusion

Specific recommendations for bicarbonates cannot be given without consideration of other soil and water constituents.

Sodium

The presence of relatively high concentration of sodium in irrigation waters affects irrigated crops in many ways. In addition to its effect on soil structure and permeability, sodium has been found by Lilleland et al. (1945)[322] and Ayers et al. (1952)[311] to be absorbed by plants and cause leaf burn in almonds, avocados, and in stone fruits grown in culture solutions. Bernstein (1967)[312] has indicated that water having SAR* values of four to eight may injure sodium-sensitive plants. It is difficult to separate the specific toxic effects of sodium from the effect of adsorbed sodium on soil structure. (This factor will be discussed later.)

As has been noted, the complex interactions of the total and relative concentrations of these common ions upon various crops preclude their consideration as individual components for general irrigation use, except for sodium and possibly chlorides in areas where fruit would be important.

Nitrate

The presence of nitrate in natural irrigation waters may be considered an asset rather than a liability with respect to plant growth. Concentrations high enough to adversely affect plant growth or composition are seldom, if ever, found. In arid regions, high nitrate water may result in nitrate accumulations in the soil in much the same manner as salt accumulates. The same soil and water management practices that minimize salt accumulation will also minimize nitrate accumulation. There is some concern over the high nitrate content of food and feed crops. Many factors such as plant species characteristics, climate conditions, and growth stage are just as significant in determining nitrate accumulations in plants as the amount present in the soil. It is unlikely that any nitrate added in natural irrigation water could be a significant factor.

Problems may arise where waste waters containing relatively large amounts of nitrogenous materials are used for irrigation. Larger amounts are usually applied than that actually required for plant growth. These wastes, however, usually contain nitrogen in a form that is slowly converted to nitrate. Nevertheless, it is possible that high nitrate accumulations in plants may occur although little evidence is available to indicate this.

Conclusion

Since nitrate in natural irrigation waters is usually an asset for plant growth and there is little evidence indicating that it will accumulate to toxic levels in irrigated plants consumed by animals, there appears to be no need for a recommendation.

Effects on Soils

Sodium Hazard Sodium in irrigation water may become a problem in the soil solution as a component of total salinity, which can increase the osmotic concentration, and as a specific source of injury to fruits. The problems of sodium mainly occur in soil structure, infiltration, and permeability rates. Since good drainage is essential for management of salinity in irrigation and for reclamation of saline lands, good soil structure and permeability must be maintained. A high percentage of exchangeable sodium in a soil containing swelling-type clays results in a dispersed condition, which is unfavorable for water movement and plant growth. Anything that alters the composition of the soil solution, such as irrigation or fertilization, disturbs the equilibrium and alters the distribution of adsorbed ions in the soil. When calcium is the predominant cation adsorbed on the soil exchange complex, the soil tends to have a granular structure that is easily worked and readily permeable. When the amount of adsorbed sodium exceeds 10 to 15 per cent of the total cations on the exchange complex, the clay becomes dispersed and slowly permeable, unless a high concentration of total salts causes flocculation. Where soils have a high exchangeable sodium content and are flocculated because of the presence of free salts in solution, subsequent removal of salts by leaching will cause sodium dispersal, unless leaching is accomplished by adding calcium or calcium-producing amendments.

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* For definition of SAR, Sodium Adsorption Ratio, see p. 330.

Adsorption of sodium from a given irrigation water is a function of the proportion of sodium to divalent cations (calcium and magnesium) in that water. To estimate the degree to which sodium will be adsorbed by a soil from a given water when brought into equilibrium with it, the Salinity Laboratory (1954)[335] proposed the sodium adsorption ratio (SAR):

$$\frac{Na^{+}}{\sqrt{\frac{Ca^{++}+Mg^{++}}{2}}} \qquad \text{Expressed as me/l}$$

As soils tend to dry, the SAR value of the soil solution increases even though the relative concentrations of the cations remain the same. This is apparent from the SAR equation, where the denominator is a square-root function. This is a significant factor in estimating sodium effects on soils.

The SAR value can be related to the amount of exchangeable cation content. This latter value is called the exchangeable sodium percentage (ESP). From empirical determinations, the U. S. Salinity Laboratory (1954)[335] obtained an equation for predicting a soil ESP value based on the SAR value of a water in equilibrium with it. This is expressed as follows:

$$ESP=\frac{[100\ a+b(SAR)]}{[1+a+b(SAR)]}$$

The constants "a" (intercept representing experimental error) and "b" (slope of the regression line) were determined statistically by various investigators who found "a" to be in the order of −0.06 to 0.01 and "b" to be within the range of 0.014 to 0.016. This relationship is shown in the nomogram (Figure V-4) developed by the U. S. Salinity Laboratory (1954).[335] For sensitive fruits, the tolerance limit for SAR of irrigation water is about four. For general crops, a limit of eight to 18 is generally considered within a usable range, although this depends to some degree on the type of clay mineral, electrolyte concentration in the water, and other variables.

The ESP value that significantly affects soil properties varies according to the proportion of swelling and nonswelling clay minerals. An ESP of 10 to 15 per cent is considered excessive, if a high percentage of swelling clay minerals such as montmorillonite are present. Fair crop growth of alfalfa, cotton, and even olives, have been observed in soils of the San Joaquin Valley (California) with ESP values ranging from 60 to 70 percent (Schoonover 1963).[336]

Prediction of the equilibrium ESP from SAR values of irrigation waters is complicated by the fact that the salt content of irrigation water becomes more concentrated in the soil solution. According to the U. S. Salinity Laboratory (1954),[335] shallow ground waters 10 times as saline as the irrigation waters may be found within depths of 10 feet, and a salt concentration two to three times that of irrigation water may be reasonably expected in the first-foot depth. Under conditions where precipitation of salts and rainfall may be neglected, the salt content of irrigation water will increase to higher concentrations in the soil solution without change in relative composition. The SAR increases in proportion to the square root of the concentration; therefore, the SAR applicable for calculating equilibrium ESP in the upper root zone may be assumed to be two to three times that of the irrigation water.

Recommendation

To reduce the sodium hazard in irrigation water for a specific crop, it is recommended that the SAR value be within the tolerance limits determined by the U.S. Soil Salinity Laboratory Staff.

Biochemical Oxygen Demand (BOD) and Soil Aeration

The need for adequate oxygen in the soil for optimum plant growth is well recognized. To meet the oxygen requirement of the plant, soil structure (porosity) and soil water contents must be adequate to permit good aeration. Conditions that develop immediately following irrigation are not clearly understood.

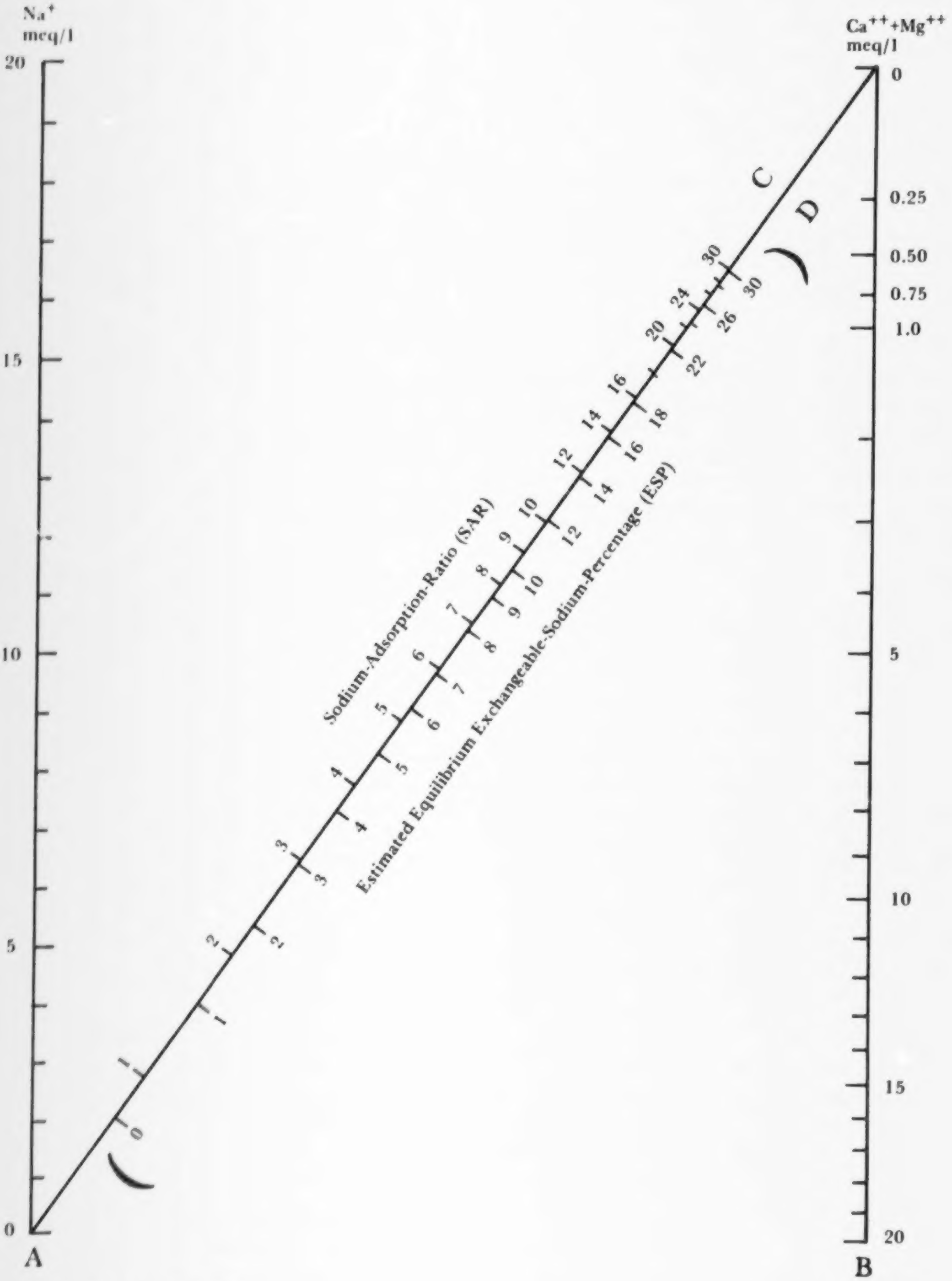
Soil aeration and oxygen availability normally present no problem on well-structured soils with good quality water. Where drainage is poor, oxygen may become limiting. Utilization of waters having high BOD or Chemical Oxygen Demand (COD) values could aggravate the condition by further depleting available oxygen. Aside from detrimental effects of oxygen deficiency for plant growth, reduction of elements such as iron and manganese to the more soluble divalent forms may create toxic conditions. Other biological and chemical equilibria may also be affected.

There is very little information regarding the effect of using irrigation waters with high BOD values on plant growth. Between source of contamination and point of irrigation, considerable reduction in BOD value may result. Sprinkler irrigation may further reduce the BOD value of water. Infiltration into well-drained soils can also decrease the BOD value of the water without serious depleting the oxygen available for plant growth.

Acidity and Alkalinity

The pH of normal irrigation water has little direct significance. Since water itself is unbuffered, and the soil is a buffered system (except for extremely sandy soils low in organic matter), the pH of the soil will not be significantly affected by application of irrigation water. There are, however, some extremes and indirect effects.

Water having pH values below 4.8 applied to acid soils over a period of time may possibly render soluble iron,



Salinity Laboratory 1954 335

FIGURE V-4—Nomogram for Determining the SAR Value of Irrigation Water and for Estimating the Corresponding ESP Value of a Soil That is at Equilibrium with the Water

aluminum, or manganese in concentrations large enough to be toxic to plant growth. Similarly, additions of saline waters to acid soils could result in a decrease in soil pH and an increase in the solubility of aluminum and manganese. In some areas where acid mine drainage contaminates water sources, pH values as low as 1.8 have been reported. Waters having unusually low pH values such as this would be strongly suspect of containing toxic quantities of certain heavy metals or other elements.

Waters having pH values in excess of 8.3 are highly alkaline and may contain high concentrations of sodium, carbonates, and bicarbonates. These constituents affect soils and plant growth directly or indirectly, (see "Effects on Plant Growth" above).

Recommendation

Because most of the effects of acidity and alkalinity in irrigation waters on soils and plant growth are indirect, no specific pH values can be recommended. However, water with pH values in the range of 4.5 to 9.0 should be usable provided that care is taken to detect the development of harmful indirect effects.

Suspended Solids

Deposition of colloidal particles on the soil surface can produce crusts that inhibit water infiltration and seedling emergence. This same deposition and crusting can reduce soil aeration and impede plant development. High colloidal content in water used for sprinkler irrigation could result in deposition of films on leaf surfaces that could reduce photosynthetic activity and thereby deter growth. Where sprinkler irrigation is used for leafy vegetable crops such as lettuce, sediment may accumulate on the growing plant affecting the marketability of these products.

In surface irrigation, suspended solids can interfere with the flow of water in conveyance systems and structures. Deposition of sediment not only reduces the capacity of these systems to carry and distribute water but can also decrease reservoir storage capacity. For sprinkler irrigation, suspended mineral solids may cause undue wear on irrigation pumps and sprinkler nozzles (as well as plugging up the latter), thereby reducing irrigation efficiency.

Soils are specifically affected by deposition of these suspended solids, especially when they consist primarily of clays or colloidal material. These cause crust formations that reduce seedling emergence. In addition, these crusts reduce infiltration and hinder the leaching of saline soils. The scouring action of sediment in streams has also been found to affect soils adversely by contributing to the dissolution and increase of salts in some areas (Pillsbury and Blaney 1966).[331] Conversely, sediment high in silt may improve the texture, consistency, and water-holding capacity of a sandy soil.

Effect on Animals or Humans

The effects of irrigation water quality on soils and plants has been discussed. However, since the quality of irrigation water is variable and originates from different sources, there may be natural or added substances in the water which pose a hazard to animals or humans consuming irrigated crops. These substances may be accumulated in certain cereals, pasture plants, or fruit and vegetable crops without any apparent injury. Of concern, however, is that the concentration of these substances may be toxic or harmful to humans or animals consuming the plants. Many substances in irrigation waters such as inorganic salts and minerals, pesticides, human and animal pathogens have recommendations to protect the desired resource. For radionuclides no such recommendation exists.

Radionuclides

There are no generally accepted standards for control of radioactive contamination in irrigation water. For most radionuclides, the use of federal Drinking Water Standards, should be reasonable for irrigation water.

The limiting factor for radioactive contamination in irrigation is its transfer to foods and eventual intake by humans. Such a level of contamination would be reached long before any damage to plants themselves could be observed. Plants can absorb radionuclides from irrigation water in two ways: direct contamination of foliage through sprinkler irrigation, and indirectly through soil contamination. The latter presents many complex problems since eventual concentration in the soil will depend on the rate of water application, the rate of radioactive decay, and other losses of the radionuclide from the soil. Some studies, relating to these factors have been reported (Menzel et al. 1963,[326] Moorby and Squire 1963,[328] Perrin 1963,[329] Menzel 1965,[325] Milbourn and Taylor 1965[327]).

It is estimated that concentrations of strontium-90 and radium-226 in fresh produce would approximate those in the irrigation water for the crop if there was negligible uptake of these radionuclides from the soil. With flood or furrow irrigation only, one or more decades of continuous irrigation with contaminated water would be required before the concentrations of strontium-90 or radium-226 in the produce equalled those in the water (Menzel *personal communication* 1972).[339]

Recommendation

In view of the lack of experimental evidence concerning the long-term accumulation and availability of strontium-90 and radium-226 in irrigated soils and to provide an adequate margin of safety it is recommended that Federal Drinking Water Standards be used for irrigation water.

SPECIFIC IRRIGATION WATER CONSIDERATIONS

Irrigation Water Quality for Arid and Semiarid Regions

Climate. Climatic variability exists in arid and semiarid regions. There can be heavy winter precipitation, generally increasing from south to north and increasing with elevation. Summer showers are common, increasing north and east from California. Common through the western part of the country is the inadequacy of precipitation during the growing season. In most areas of the West, intensive agriculture is not possible without irrigation. Irrigation must supply at least one-half of all the soil water required annually for crops for periods ranging from three to 12 months.

Annual precipitation varies in the western United States from practically zero in the southwestern deserts to more than 100 inches in the upper western slope of the Pacific Northwest. The distribution of precipitation throughout the year also varies, with no rainfall during extended periods in many locales. Often the rainfall occurs during nongrowing seasons.

The amount of precipitation and its distribution is one of the principal variables in determining the diversion requirement or demand for irrigation water.

Land. Soils in the semiarid and arid regions were developed under dry climatic conditions with little leaching of weatherable minerals in the surface horizon. Consequently, these soils are better supplied with most nutrient elements. The pH of these soils varies from being slightly acidic to neutral or alkaline. The presence of silicate clay minerals of the montmorillonite and hydrous mica groups in many of these soils gives them a higher exchange capacity than those of the southeast, which contain kaolinite minerals of lower exchange capacity. However, organic matter and nitrogen contents of arid soil are usually lower. Plant deficiencies of trace elements such as zinc, iron, manganese are more frequently encountered. Because of the less frequent passage of water through arid soils, they are more apt to be saline.

The nature of the surface horizon (plow layer) and the subsoil is especially important for irrigation. During soil formation a profile can develop consisting of various horizons. The horizons consist of genetically related layers of soil or soil material parallel to the land surface, and they differ in their chemical, physical, and biological properties. The productivity of a soil is largely determined by the nature of these horizons. Soils available for irrigation with restrictive or impervious horizons present management problems (e.g., drainage, aeration, salt accumulation in root zone, changes in soil structure) and consequently are not the best for irrigated agriculture.

Other land and soil factors of importance to irrigation are topography and slope, which may influence the choice of irrigation method, and soil characteristics. The latter are extremely important because they determine the usable depth of water that can be stored in the root zone of the crop and the erodability and intake rate of the soil.

Water. Each river system within the arid and semiarid portion of the United States has quality characteristics peculiar to its geologic origin and climatic environment. In considering water quality characteristics as related to irrigation, both historic and current data for the stream and location in question should be used with care because of the large seasonal and sporadic variations that occur.

The range of sediment concentrations of a river throughout the year is usually much greater than the range of dissolved solids concentrations. Maximum sediment concentrations may range from 10 to more than a thousand times the minimum concentrations. Usually, the sediment concentrations are higher during high flow than during low flow. This differs inversely from dissolved-solids concentrations that are usually lower during high flows.

Four general designations of water have been used (Rainwater 1962)[361] based on their chemical composition: (1) calcium-magnesium, carbonate-bicarbonate; (2) calcium-magnesium, sulfate-chloride; (2) sodium-potassium, carbonate-bicarbonate; and (4) sodium-potassium, sulfate-chloride. This type of classification characterizes the chemical properties of the water and would be indicative of reactions that could be expected with soil when used for irrigation. Although a listing of data for each stream and tributary is beyond the scope of this report, an indication of ranges in dissolved-solids concentrations, chemical type, and sediment concentration is given in Table V-10 (Rainwater 1962).[361]

Customarily, each irrigation project diverts water at one point in the river and the return flow comes back into the mainstream somewhere below the system. This return flow consists in the main of (1) regulatory water, which is the unused part of the diverted water required so that each farmer irrigating can have the exact flow he has ordered;

TABLE V-10—Variations in Dissolved Solids, Chemical Type, and Sediment in Rivers in Arid and Semiarid United States

Region	Dissolved solids concentrations, mg/l		Prevalent chemical type[a]	Sediment concentrations, mg/l[b]	
	From	To		From	To
Columbia River Basin	<100	300	Ca-Mg, C-b	<200	300
Northern California	<100	700	Ca-Mg, C-b	<200	+500
Southern California	<100	+2,000	Ca-Mg, C-b; Ca-Mg, S-C	<200	+15,000
Colorado River Basin	<100	+2,500	Ca-Mg, S-C; Ca-Mg, C-b	<200	+15,000
Rio Grande Basin	<100	+2,000	Ca-Mg, C-b; Ca-Mg, S-C	+300	+50,000
Pecos River Basin	100	+3,000	Ca-Mg, S-C	+300	+7,000
Western Gulf of Mexico Basins	100	+3,000	Ca-Mg, C-b; Ca-Mg, S-C; Na-P, S-C	<200	+30,000
Red River Basin	<100	+2,500	Ca-Mg, S-C; Na-P, S-C	+300	+25,000
Arkansas River Basin	100	+2,000	Ca-Mg, S-C; Ca-Mg, C-b; Na-P, S-C	+300	+30,000
Platte River	100	+1,500	Ca-Mg, C-b; Ca-Mg, S-C	+300	+7,000
Upper Missouri River Basin	100	+2,000	Ca-Mg, S-C; Na-P, C-b; Na-P, C-b	<200	+15,000

[a] Ca-Mg, C-b = Calcium-magnesium, carbonate-bicarbonate. Ca-Mg, S-C = Calcium-magnesium, sulfate-chloride. Na-P, C-b = Sodium-potassium, carbonate-bicarbonate. Na-P, S-C = Sodium-potassium, sulfate-chloride.

[b] $\text{Sediment concentration} = \frac{\text{Annual Load}}{\text{Annual Streamflow}}$

Rainwater 1962[361].

(2) tail water, which is that portion of the water that runs off the ends of the fields; and (3) underground drainage, required to provide adequate application and salt balance in all parts of the fields. The initial flush of tail water may be somewhat more saline than later but rapidly approaches the same quality as the applied water (Reeve et al. 1955).[362]

Drainage and Leaching Requirements. In all irrigation agriculture some water must pass through the soil to remove salts brought to the soil in the water. In semiarid areas, or in the transition zone between arid and humid regions, this drainage water is usually obtained as a result of rainfall during periods of low evapotranspiration, and no excess irrigation water is needed to provide the drainage required. In many arid regions, the needed leaching must be obtained by adding excess water. In all cases, the required drainage volume is related to the amount of salt in the irrigation water. That drainage volume is called the leaching requirement (LR).

It is possible to predict the approximate salt concentration that would occur in the soil after a number of irrigations by estimating the proportion of applied water that will percolate below the root zone. In any steady-state leaching formula, the following assumptions are made:

- No precipitation of salts occurs in the soil;
- Ion uptake by plants is negligible;
- There is uniform distribution of soil moisture through the profile and uniform concentration of salts in the soil moisture;
- Complete and uniform mixing of irrigation water with soil moisture takes place before any of the moisture percolates below the root zone and
- Residual soil moisture is negligible.

A steady state leaching requirement formula has been developed by the U.S. Salinity Laboratory (1954)[363] designed to estimate that fraction of the irrigation water that must be leached through the root zone to control soil salinity at any specified level. This is given as:

$$\mathrm{LR} = \frac{D_{dw}}{D_{iw}} = \frac{EC_{iw}}{EC_{dw}}$$

where LR is the leaching requirement; D_{dw}, the depth of drainage water; D_{iw}, the depth of irrigation water; EC_{iw}, the salinity of irrigation water; and EC_{dw}, the salinity of water percolating past root zone.

Hence, if EC_{dw} is determined by the salt tolerance of the crop to be grown, and the salt content of the irrigation water EC_{iw} is known, the desired LR can be calculated. This leaching fraction will then be the ratio of depth of drainage volume to the depth of irrigation water applied.

Because the permissible values for EC_{dw} for various yield decrements for various crops are not known, the EC_e for 50 per cent yield reduction has been substituted for EC_{dw}. The actual yield reduction will probably be less than 50 per cent (Bernstein 1966).[340] This EC_e is the assumed average electrical conductivity for the soil water at saturation for the whole root zone. When it is substituted for the EC_{dw}, the actual EC_e encountered in the root zone will be less than this value because, in many near steady state situations, the salinity increases progressively with increase in depth in the profile and is maximum at the bottom of the root zone.

Bernstein (1967)[341] has developed a leaching fraction formula that takes into consideration factors that control leaching rates such as infiltration rate, climate (evapotranspiration), frequency and duration of irrigation, and, of course, the salt tolerance of the crops. He defines the leaching fraction as $\mathrm{LF} = 1 - ET_e/IT_1$ where LF is the leaching fraction or proportion of applied water percolating below the root zone; E, the average rate of evapotranspiration during the irrigation cycle, T_e; and I, the average infiltration rate during the period of infiltration, T_1. By utilizing both the required leaching derived from the steady state formula

$$\mathrm{LR} = \frac{EC_{iw}}{EC_{dw}}$$

and the leaching fraction based upon infiltration rates and evapotranspiration during the irrigation cycle, it is possible to estimate whether adequate leaching can be attained or whether adjustments must be made in the crops to be grown to permit higher salinity concentrations.

In addition to determination of crops to be grown, leaching requirements may be used to indicate the total quantities of water required. For example, irrigation water with a conductivity of two mmhos requires one-sixth more water to maintain root zone salt concentrations within eight mmhos than would water with a salt concentration of one mmhos under the same conditions of use.

There are a number of problems in applying the leaching requirement concept in actual practice. Some of these relate to the basic assumptions involved and others derive from water application problems and soil variability.

- Considerable precipitation of calcium carbonate occurs as many irrigation waters enter the soil causing a reduction in the total soluble salt load. In many crops, or crop rotations, crop removal of such ions as chloride was a significant fraction of the total added in waters of medium to low salinity. (Pratt et al. 1967)[359]
- It is not practical to apply water with complete uniformity.
- Soils are far from uniform, particularly with respect to vertical hydraulic conductivity.
- The effluent from tile or ditch drains may not be representative of the salinity of water at the bottom of the root zones.

Also, there is a considerable variation in drainage outflow that has no relation to leaching requirement when different crops are irrigated (Pillsbury and Johnston 1965).[357] This results from variations in irrigation practices for the different crops.

The leaching requirement concept, while very useful, should not be used as a sole guide in the field. The leaching requirement is a long-period average value that can be departed from for short periods with adequately drained soils to make temporary use of water poorer in quality than customarily applied.

The exact manner in which leaching occurs and the appropriate values to be used in leaching requirement formulas require further study. The many variables and assumptions involved preclude a precise determination under field conditions.

Salinity Hazard. Waters with total dissolved solids (TDS) less than about 500 mg/l are usually used by farmers without awareness of any salinity problem, unless, of course, there is a high water table. Also, without dilution from precipitation or an alternative supply, waters with TDS of about 5,000 mg/l usually have little value for irrigation (Pillsbury and Blaney 1966).[356] Within these limits, the *value* of the water appears to decrease as the salinity increases. Where water is to be used regularly for the irrigation of relatively impervious soil, its value is limited if the TDS is in the range of 2,000 mg/l or higher.

Recommendation

In spite of the facts that (1) any TDS limits used in classifying the salinity hazard of waters are somewhat arbitrary; (2) the hazard is related not only to the TDS but also to the individual ions involved; and (3) no exact hazard can be assessed unless the soil, crop, and acceptable yield reductions are known, Table V-11 suggests classifications for general purposes for arid and semiarid regions.

TABLE V-11—Recommended Guidelines for Salinity in Irrigation Water

Classification	TDS mg/l	EC mmhos/cm
Water for which no detrimental effects are usually noticed	500	0.75
Water that can have detrimental effects on sensitive crops	500–1,000	0.75–1.50
Water that can have adverse effects on many crops; requires careful management practices	1,000–2,000	1.50–3.00
Water that can be used for tolerant plants on permeable soils with careful management practices	2,000–5,000	3.00–7.50

Permeability Hazard. Two criteria used to evaluate the effect of salts in irrigation water on soil permeability are: (1) the sodium adsorption ratio (SAR) and its relation to the exchangeable sodium percentage, and (2) the bicarbonate hazard that is particularly applicable to waters of arid regions. Another factor related to the permeability hazard is that the permeability tends to increase, and the stability of a soil at any exchangeable sodium percentage (ESP) increases as the salinity of the water increases (Quirk and Schofield 1955).[360]

Eaton (1950),[317] Doneen (1959),[346] and Christiansen and Thorne (1966)[345] have recognized that the permeability hazard of irrigation waters containing bicarbonate was greater than indicated by their SAR values. Bower and Wilcox (1965)[343] found that the tendency for calcium carbonate to precipitate in soils was related to the Langelier index (Langelier 1936)[349] and to the fraction of the irrigation water evapotranspired from the soil. Bower et al. (1965,[344] 1968)[342] modified the Langelier index or precipitation index (PI) to the soil system and presented simplified means for calculation. The PI was 8.4-pH$_c$, where 8.4 was the pH of the soil and pH$_c$, the pH that would be found in a calcium carbonate suspension that would have the same calcium and bicarbonate concentrations as those in the irrigation water. For the soil system

$$pH_c = pK_2 - pK_c + p(Ca + Mg) + pAlk$$

where pK$_2$ and pK$_c$ are the negative logarithms, respectively, of the second dissociation constant for carbonic acid and the solubility constant for calcite; p(Ca+Mg) and pAlk are the negative logarithms, respectively, of the molar concentrations of (Ca+Mg) and the titrable alkalinity. Magnesium is included primarily because it reacts, through cation exchange, to maintain the calcium concentration in solution. The PI combines empirically with the SAR in the following equation

$$SAR_{se} = SAR_{iw}\ \sqrt{C}(1 + PI)$$

where SAR$_{se}$ and SAR$_{iw}$ are for the saturation extract and the irrigation water, respectively, C is the concentration factor or the reciprocal of the leaching fraction, and PI is 8.4-pH$_c$. Bower et al. (1968)[342] and Pratt and Bair (1969),[358] using lysimeter experiments, have shown a high correlation between the predicted and measured SAR$_{se}$ with waters of various bicarbonate concentrations. The information available suggested a high utility of this equation for calculating permeability or sodium hazard of waters. In cases where C is not known, a value of 4, corresponding to a leaching fraction of 0.25, can be used to give relative comparisons among waters. In this case the equation is

$$SAR_{se} = 2SAR_{iw}(1 + PI).$$

Data can be used to prepare graphs, from which the values for pK$_2$−pK$_c$, p(Ca+Mg), and pAlk can be obtained for easy calculation of pH$_c$. The calculation of pH$_c$ is described by Bower et al. (1965).[344]

Soils have individual responses in reduction in permeability as the SAR or calculated SAR values increase, but adverse effects usually begin to appear as the SAR value passes through the range from 8 to 18. Above an SAR of 18 the effects are usually adverse.

Suspended Solids. Suspended organic solids in surface water supplies seldom give trouble in ditch distribution systems except for occasional clogging of gates. They can also carry weed seeds onto fields where their subsequent growth can have a severely adverse effect on the crop or can have a beneficial effect by reducing seepage losses. Where surface water supplies are distributed through pipelines, it is often necessary to have self-cleaning screens to prevent clogging of the pipe system appliances. Finer screening is usually required where water enters pressure-pipe systems for sprinkler irrigation.

There are waters diverted for irrigation that carry heavy inorganic sediment loads. The effects that these loads might have depend in part on the particle size and distribution of the suspended material. For example, the ability of sandy soils to store moisture is greatly improved after the soils are irrigated with muddy water for a period of years. More commonly, sediment tends to fill canals and ditches, causing serious cleaning and dredging problems. It also tends to further reduce the already low infiltration characteristics of slowly permeable soils.

Irrigation Water Quality For Humid Regions

Climate The most striking feature of the climate of the humid region that contrasts with that of the far West and intermountain areas is the larger amount of and less seasonable distribution of the precipitation. Abundant rainfall, rather than lack of it, is the normal expectation. Yet, droughts are common enough to require that attention be given to supplemental irrigation. These times of shortage of water for optimum plant growth can occur at irregular intervals and at almost any stage of plant growth.

Water demands per week or day are not as high in humid as in arid lands. But rainfall is not easily predicted. Thus a crop may be irrigated and immediately thereafter receive a rain of one or two inches. Supplying the proper amount of supplemental irrigation water at the right time is not easy even with adequate equipment and a good water supply. There can be periods of several successive years when supplemental irrigation is not required for most crops in the humid areas. There are times however, when supplemental water can increase yield or avert a crop failure. Supplemental irrigation for high-value crops will undoubtedly increase in humid areas in spite of the fact that much capital is tied up in irrigation equipment during years in which little or no use is made of it.

The range of temperatures in the humid region in which supplemental irrigation is needed is almost as great as that for arid and semiarid areas. It ranges from that of the short growing season of upstate New York and Michigan to the continuous growing season of southern Florida. But in the whole of this area, the most unpredictable factor in crop production is the need for additional water for optimum crop production.

Soils The soils of the humid region contrast with those of the West primarily in being lower in available nutrients. They are generally more acid and may have problems with exchangeable aluminum. The texture of soils is similar to that found in the West and ranges from sands to clays. Some are too permeable, while others take water very slowly.

Soils of the humid region generally have clay minerals of lower exchange capacity than soils of the arid and semiarid regions and hence lower buffer capacity. They are more easily saturated with anions and cations. This is an important consideration if irrigation with brackish water is necessary to supplement natural rainfall. Organic matter content ranges from practically none on some of the Florida sands to 50 per cent or more in irrigated peats.

One of the most important characteristics of many of the soils of the humid Southeast is the unfavorable root environment of the deeper horizons containing exchangeable aluminum and having a strong acid reaction. In fact, the lack of root penetration of these horizons by most farm crops is the primary reason for the need for supplemental irrigation during short droughts.

Specific Difference Between Humid and Arid Regions The effect of a specific water quality deterrent on plant growth is governed by related factors. Basic principles involved are almost universally applicable, but the ultimate effect must take into consideration these associated variables. Water quality criteria for supplemental irrigation in humid areas may differ from those indicated for arid and semiarid areas where the water requirements of the growing plant are met almost entirely by irrigation.

When irrigation water containing a deterrent is used, its effect on plant growth may vary, however, with the stage of growth at which the water is applied. In arid areas, plants may be subjected to the influence of irrigation water quality continuously from germination to harvest. Where water is used for supplemental irrigation only, the effect on plants depends not only upon the growth stage at which applied, but to the length of time that the deterrent remains in the root zone (Lunin et al. 1963).[352] Leaching effects of intervening rainfall must be taken into consideration.

Climatic differences between humid and arid regions also influence criteria for use of irrigation water. The amount of rainfall determines in part the degree to which a given constituent will accumulate in the soil. Other factors associated with salt accumulation in the soil are those climatic conditions relating to evapotranspiration. In humid areas, evapotranspiration is generally less than in arid regions, and plants are not as readily subjected to water stress. The importance of climatic conditions in relation to salinity was demonstrated by Magistad et al. (1943).[355] In general, criteria regarding salinity for supplemental irrigation in humid areas can be more flexible than for arid areas.

Soil characteristics represent another significant difference between arid and humid regions. These were discussed previously.

Mineralogical composition will also vary. The composition of soil water available for absorption by plant roots represents the results of an interaction between the constituents of the irrigation water and the soil complex. The final result may be that a given quality deterrent present in the water could be rendered harmless by the soil (remaining readily available), or that the dissolved constituents of a water may render soluble toxic concentrations of an element that was not present in the irrigation water. An example of this would be the addition of a saline water to an acid soil resulting in a decrease in pH and a possible increase in solubility of elements such as iron, aluminum, and manganese (Eriksson 1952).[348]

General relationships previously derived for SAR and adsorbed sodium in neutral or alkaline soils of arid areas do not apply equally well to acid soils found in humid regions (Lunin and Batchelder 1960).[350] Furthermore, the effect of a given level of adsorbed sodium (ESP) on plant growth is determined to some degree by the associated adsorbed cations. The amount of adsorbed calcium and magnesium relative to adsorbed sodium is of considerable consequence, especially when comparing acidic soils to ones that are neutral or alkaline. Another example would be the presence of a trace element in the irrigation water that might be rendered insoluble when applied to a neutral or alkaline soil, but retained in a soluble, available form in acid soils. For these reasons, soil characteristics, which differ greatly between arid and humid areas, must be taken into consideration.

Certain economic factors also influence water quality criteria for supplemental irrigation. Although the ultimate objective of irrigation is to insure efficient and economic crop production, there may be instances where an adequate supply of good quality water is unavailable to achieve this. A farmer may be faced with the need to use irrigation water of inferior quality to get some economic return and prevent a complete crop failure. This can occur in humid areas during periods of prolonged drought. Water quality criteria are generally designed for optimum production, but consideration must be given also to supplying guidelines for use of water of inferior quality to avert a crop failure.

Specific Quality Criteria for Supplemental Irrigation A previous discussion (see "Water Quality Considerations for Irrigation" above) of potential quality deterrents contained a long list of factors indicating the current state of our knowledge as to how they might relate to plant growth. Criteria can be established by determining a concentration of a given deterrent, which, when adsorbed on or absorbed by a leaf during sprinkler irrigation, results in adverse plant growth, and by evaluating the direct or indirect effects (or both) that a given concentration of a quality deterrent has on the plant root environment as irrigation water enters the soil. Neither evaluation is simple, but the latter is more complex because so many variables are involved. Since sprinkler application in humid areas is most common for supplemental irrigation, both types of evaluation have considerable significance. The following discussion relates only to those quality criteria that are specifically applicable to supplemental irrigation.

Salinity. General concepts regarding soil salinity as previously discussed are applicable. Actual levels of salinity that can be tolerated for supplemental irrigation must take into consideration the leaching effect of rainfall and the fact that soils are usually nonsaline at spring planting. The amount of irrigation water having a given level of salinity that can be applied to the crop will depend upon the number of irrigations between leaching rains, the salt tolerance of the crop, and the salt content of the soil prior to irrigation.

Since it is not realistic to set a single salinity value or even a range that would take these variables into consideration, a guide was developed to aid farmers in safely using saline or brackish waters (Lunin and Gallatin 1960).[351] The following equation was used as a basis for this guide:

$$EC_{e(f)} = EC_{e(i)} + \frac{n(EC_{iw})}{2}$$

where $EC_{e(f)}$ is the electrical conductivity of the saturation extract after irrigation is completed; $EC_{e(i)}$, the electrical conductivity of the soil saturation extract before irrigation; EC_{iw}, the electrical conductivity of the irrigation water; and n, the number of irrigations.

To utilize this guide, the salt tolerance of the crop to be grown and the soil salinity level ($EC_{e(f)}$) that will result in a 15 or 50 per cent yield decrement for that crop must be considered. After evaluating the level of soil salinity prior to irrigation ($EC_{e(i)}$) and the salinity of the irrigation water, the maximum number of permissible irrigations can be calculated. These numbers are based on the assumption that no intervening rainfall occurs in quantities large enough to leach salts from the root zone. Should leaching rainfall occur, the situation could be reevaluated using a new value for $EC_{e(i)}$.

Categorizing the salt tolerance of crops as highly salt tolerant, moderately salt tolerant, and slightly salt tolerant, the guide shown in Table V-12 was prepared to indicate

TABLE V-12—Permissible Number of Irrigations in Humid Areas with Saline Water between Leaching Rains for Crops of Different Salt Tolerance [a]

Irrigation water		Number of irrigations for crops having		
Total salts mg/l	Electrical conductivity mmhos/cm at 25 C	Low salt tolerance	Moderate salt tolerance	High salt tolerance
640	1	7	15	....
1,280	2	4	7	11
1,920	3	2	4–5	7
2,560	4	2	3	5
3,200	5	1	2–3	4
3,840	6	1	2	3
4,480	7	....	1–2	2–3
5,120	8	....	1	2

[a] Based on a 50 per cent yield decrement.

Lunin et al. 1960[354].

the number of permissible irrigations using water of varying salt concentrations. This guide is based on two assumptions:

- no leaching rainfall occurs between irrigations.
- there is no salt accumulation in the soil at the start of the irrigation period. If leaching rains occur between irrigations, the effect of the added salt is minimized. If there is an accumulation of salt in the soil initially, such as might occur when irrigating a fall crop on land to which saline water had been applied during a spring crop, the soil should be tested for salt content, and the irrigation recommendations modified accordingly.

Recommendation

Since it is not realistic to set a single salinity value or even a range that would take all variables into consideration, Table V–12 developed by Lunin et al. (1960),[354] should be used as a guide to aid farmers in safely using saline or brackish waters for supplemental irrigation in humid areas.

SAR values and exchangeable sodium. The principles relating to SAR values and the degree to which sodium is adsorbed from water by soils are generally applicable in both arid and humid regions. Some evidence is available (Lunin and Batchelder 1960),[350] however, to indicate that, for a given water quality, less sodium was adsorbed by an acid soil than by a base-saturated soil. For a given level of exchangeable sodium, preliminary evidence indicated more detrimental effects on acid soils than on base-saturated soils (Lunin et al. 1964).[353]

Experimental evidence is not conclusive, so the detrimental limits for SAR values listed previously should also apply to supplemental irrigation in humid regions. (See the recommendation in this section following the discussion of sodium hazard under Water Quality Considerations for Irrigation.)

Acidity and alkalinity. The only consideration not previously discussed relates to soil acidity, which is more prevalent in humid regions where supplemental irrigation is practiced. Any factor that drops the pH below 4.8 may render soluble toxic concentrations of iron, aluminum, and manganese. This might result from application of a highly acidic water or from a saline solution applied to an acidic soil. (See the recommendation in this section following the discussion of acidity and alkalinity under Water Quality Considerations for Irrigation.)

Trace elements. Criteria and related factors discussed in the section on Phytotoxic Trace Elements are equally applicable to supplemental irrigation in humid regions. Certain related qualifications must be kept in mind, however. First, foliar absorption of trace elements in toxic amounts is directly related to sprinkler irrigation. Critical levels established for soil or culture solutions would not apply to direct foliar injury. Regarding trace element concentrations in the soil resulting from irrigation water application, the volume of the water applied by sprinkler as supplemental irrigation is much less than that applied by furrow or flood irrigation in arid regions.

In assessing trace element concentrations in irrigation water, total volume of water applied and the physicochemical characteristics of the soil must be taken into consideration. Both factors could result in different criteria for supplemental irrigation as compared with surface irrigation in arid regions.

Suspended solids. Certain factors regarding suspended solids must be taken into consideration for sprinkler irrigation. The first deals with the plugging up of sprinkler nozzles by these sediments. Size of sediment is a definite factor, but no specific particle size limit can be established. If some larger sediment particles pass through the sprinkler, they can often be washed off certain leafy vegetable crops. Some of the finer fractions, suspended colloidal material, could accumulate on the leaves and, once dry, become extremely difficult to wash off, thereby impairing the quality of the product.

PHYTOTOXIC TRACE ELEMENTS

In addition to the effect of total salinity on plant growth, individual ions may cause growth reductions. Ions of both major and trace elements occur in irrigation water. Trace elements are those that normally occur in waters or soil solutions in concentrations less than a few mg/l with usual concentrations less than 100 microgram (μg)/l. Some may be essential for plant growth, while others are nonessential.

When an element is added to the soil, it may combine with it to decrease its concentration and increase the store of that element in the soil. If the process of adding irrigation water containing a toxic level of the element continues, the capacity of the soil to react with the element will be saturated. A steady state may be approached in which the amount of the element leaving the soil in the drainage water equals the amount added with the irrigation water, with no further change in concentration in the soil. Removal in harvested crops can also be a factor in decreasing the accumulation of trace elements in soils.

In many cases, soils have high capacities to react with trace elements. Therefore, irrigation water containing toxic levels of trace elements may be added for many years before a steady state is approached. Thus, a situation exists where toxicities may develop in years, decades, or even centuries from the continued addition of pollutants to irrigation waters. The time would depend on soil and plant factors as well as on the concentration of trace elements in the water.

Variability among species is well recognized. Recent investigations by Foy et al. (1965),[402] and Kerridge et al. (1971)[425] working with soluble aluminum in soils and in nutrient solutions, have demonstrated that there is also variability among varieties within a given species.

Comprehensive reviews of literature dealing with trace element effects on plants are provided by McKee and Wolf (1963),[436] Bolland and Butler (1966),[378] and Chapman (1966).[386] Hodgson (1963)[417] presented a review dealing with reactions of trace elements in soils.

In developing a workable program to determine acceptable limits for trace elements in irrigation waters, three considerations should be recognized:

- Many factors affect the uptake of and tolerance to trace elements. The most important of these are the natural variability in tolerances of plants and of animals that consume plants, in reactions within the soil, and in nutrient interactions, particularly in the plant.
- A system of tolerance limits should provide sufficient flexibility to cope with the more serious factors listed above.
- At the same time, restrictions must be defined as precisely as possible using presently available, but limited, research information.

Both the concentration of the element in the soil solution, assuming that steady state may be approached, and the total amount of the element added in relation to quantities that have been shown to produce toxicities were used in arriving at recommended maximum concentrations. A water application rate of 3 acre feet/acre/year was used to calculate the yearly rate of trace elements added in irrigation water.

The suggested maximum trace element concentrations for irrigation waters are shown in Table V-13.

The suggested maximum concentrations for continuous use on all soils are set for those sandy soils that have low capacities to react with the element in question. They are generally set at levels less than the concentrations that produce toxicities when the most sensitive plants are grown in nutrient solutions or sand cultures. This level is set, recognizing that concentration increases in the soil as water is evapotranspired, and that the effective concentration in the soil solution, at near steady state, is higher than in the irrigation water. The criteria for short-term use are suggested for soils that have high capacitites to remove from solution the element or elements being considered.

The work of Hodgson (1963)[417] showed that the general tolerance of the soil-plant system to manganese, cobalt, zinc, copper, and boron increased as the pH increased, primarily because of the positive correlation between the capacity of the soil to inactivate these ions and the pH. This same relationship exists with aluminum and probably exists with other elements such as nickel (Pratt et al. 1964)[449] and boron (Sims and Bingham 1968).[465] However, the ability of the soil to inactivate molybdenum decreases with increase in pH, such that the amount of this element that could be added without producing excesses was higher in acid soils.

TABLE V-13—Recommended Maximum Concentrations of Trace Elements in Irrigation Waters[a]

Element	For waters used continuously on all soil mg/l	For use up to 20 years on fine textured soils of pH 6.0 to 8.5 mg/l
Aluminum	5.0	20.0
Arsenic	0.10	2.0
Beryllium	0.10	0.50
Boron	0.75	2.0
Cadmium	0.010	0.050
Chromium	0.10	1.0
Cobalt	0.050	5.0
Copper	0.20	5.0
Fluoride	1.0	15.0
Iron	5.0	20.0
Lead	5.0	10.0
Lithium	2.5[b]	2.5[b]
Manganese	0.20	10.0
Molybdenum	0.010	0.050[d]
Nickel	0.20	2.0
Selenium	0.020	0.020
Tin[c]	....	....
Titanium[c]	....	....
Tungsten[c]	....	....
Vanadium	0.10	1.0
Zinc	2.0	10.0

[a] These levels will normally not adversely affect plants or soils.
[b] Recommended maximum concentration for irrigating citrus is 0.075 mg/l.
[c] See text for a discussion of these elements.
[d] For only acid fine textured soils or acid soils with relatively high iron oxide contents.

In addition to pH control (i.e., liming acid soils), another important management factor that has a large effect on the capacity of soils to adsorb some trace elements without development of plant toxicities is the available phosphorus level. Large applications of phosphate are known to induce deficiencies of such elements as copper and zinc and greatly reduce aluminum toxicity (Chapman 1966).[386]

The concentrations given in Table V-13 are for ionic and soluble forms of the elements. If insoluble forms are present as particulate matter, these should be removed by filtration before the water is analyzed.

Aluminum

The toxicity of this ion is considered to be one of the main causes of nonproductivity in acid soils (Coleman and Thomas 1967,[392] Reeve and Sumner 1970,[453] Hoyt and Nyborg 1971a[419]).

At pH values from about 5.5 to 8.0, soils have great capacities to precipitate soluble aluminum and to eliminate its toxicity. Most irrigated soils are naturally alkaline, and many are highly buffered with calcium carbonate. In these situations aluminum toxicity is effectively prevented.

With only a few exceptions, as soils become more acid ($pH < 5.5$), exchangeable and soluble aluminum develop by dissolution of oxides and hydroxides or by decomposition of clay minerals. Thus, without the introduction of aluminum, a toxicity of this element usually develops as soils are acidified, and limestone must be added to keep the soil productive.

In nutrient solutions toxicities are reported for a number of plants at aluminum concentrations of 1 mg/l (Pratt 1966),[448] whereas wheat is reported to show growth reductions at 0.1 mg/l (Barnette 1923).[370] Liebig et al. (1942)[432] found growth depressions of orange seedlings at 0.1 mg/l. Ligon and Pierre (1932)[433] showed growth reductions of 60, 22, and 13 per cent for barley, corn, and sorghum, respectively, at 1 mg/l.

In spite of the potential toxicity of aluminum, this is not the basis for the establishment of maximum concentrations in irrigation waters, because ground limestone can be added where needed to control aluminum solubility in soils. Nevertheless, two disadvantages remain. One is that the salts that are the sources of soluble aluminum in waters acidify the soil and contribute to the requirement for ground limestone to prevent the accumulation or development of soluble aluminum. This is a disadvantage only in acid soils. The other disadvantage is a greater fixation of phosphate fertilizer by freshly precipitated aluminum hydroxides.

In determining a recommendation for maximum levels of aluminum in irrigation water using 5.0 mg/l for waters to be used continuously on all soils and 20 mg/l for up to 20 years on fine-textured soils, the following was considered. At rates of 3 acre feet of water per acre per year the calcium carbonate equivalent of the 5 mg/l concentration used for 100 years would be 11.5 tons per acre; the 20 mg/l concentration for 20 years would be equivalent to 9 tons of $CaCO_3$ per acre. In most irrigated soils this amount of limestone would not have to be added, because the soils have sufficient buffer capacity to neutralize the aluminum salts. In acid soils that are already near the pH where limestone should be used, the aluminum added in the water would contribute these quantities to the lime requirements.

Amounts of limestone needed for control of soluble aluminum in acid soils can be estimated by a method that is based on pH control (Shoemaker et al. 1961).[463] A method based on the amount of soluble and exchangeable aluminum was developed by Kamprath (1970).[424]

Recommendations

Recommended maximum concentrations are 5.0 mg/l aluminum for continuous use on all soils and 20 mg/l for use on fine textured neutral to alkaline soils over a period of 20 years.

Arsenic

Albert and Arndt (1931)[368] found that arsenic at 0.5 mg/l in nutrient solutions reduced the growth of roots of cowpeas, and at 1.0 mg/l it reduced the growth of both roots and tops. They reported that 1.0 mg/l of soluble arsenic was frequently found in the solution obtained from soils with demonstrated toxic levels of arsenic. Rasmussen and Henry (1965)[451] found that arsenic at 0.5 mg/l in nutrient solutions produced toxicity symptoms in seedlings of the pineapple and orange. Below this concentration no symptoms of toxicity were found. Clements and Heggeness (1939)[390] reported that 0.5 mg/l arsenic as arsenite in nutrient solutions produced an 80 per cent yield reduction in tomatoes. Liebig et al. (1959)[431] found that 10 mg/l of arsenic as arsenate or 5 mg/l as arsenite caused marked reduction in growth of tops and roots of citrus grown in nutrient solutions. Machlis (1941)[434] found that concentrations of 1.2 and 12 mg/l caused growth suppression in beans and sudan grass respectively.

However, the most definite work with arsenic toxicity in soils has been aimed at determining the amounts that can be added to various types of soils without reduction in yields of sensitive crops. The experiments of Cooper et al. (1932),[393] Vandecaveye et al. (1936),[472] Crafts and Rosenfels (1939),[394] Dorman and Colman (1939),[396] Dorman et al. (1939),[397] Clements and Munson (1947),[391] Benson (1953),[372] Chisholm et al. (1955),[388] Jacobs et al. (1970),[422] Woolson et al. (1971)[481] showed that the amount of total arsenic that produced the initiation of toxicity varied with soil texture and other factors that influenced the adsorptive capacity. Assuming that the added arsenic is mixed with the surface six inches of soil and that it is in the arsenate form, the amounts that produce toxicity for sensitive plants vary from 100 pounds (lb)/acre for sandy soils to 300 lb/acre for clayey soils. Data from Crafts and Rosenfels (1939)[394] for 80 soils showed that for a 50 per cent yield reduction with barley, 120, 190, 230, and 290 lb arsenic/acre were required for sandy loams, loams, clay loams, and clays, respectively. These amounts of arsenic indicated the amounts adsorbed into soils of different adsorptive capacities before the toxicity level was reached.

With long periods of time involved, such as would be the case with accumulations from irrigation water, possible leaching in sandy soils (Jacobs et al. 1970)[422] and reversion to less soluble and less toxic forms of arsenic (Crafts and Rosenfels 1939)[394] allow extensions of the amounts required for toxicity. Perhaps a factor of at least two could be used, giving a limit of 200 lb in sandy soils and a limit of 600 lb in clayey soils over many years. Using these limits, a concentration of 0.1 mg/l could be used for 100 years on sandy soils, and a concentration of 2 mg/l used for a period of 20 years or 0.5 mg/l used for 100 years on clayey soils would provide an adequate margin of safety. This is assuming 3 acre feet of water are used per acre per year (1 mg/l equals 2.71 lb/acre foot of water or 8.13 lb/3 acre feet), and that the added arsenic becomes mixed in a 6-inch layer of soil. Removal of small amounts in harvested crops provides an additional safety factor.

The only effective management practice known for soils that have accumulated toxic levels of arsenic is to change to more tolerant crops. Benson and Reisenauer (1951)[373] developed a list of plants of three levels of tolerance. Work by Reed and Sturgis (1936)[452] suggested that rice on flooded soils was extremely sensitive to small amounts of arsenic, and that the suggested maximum concentrations listed below were too high for this crop.

Recommendations

Recommendations are that maximum concentrations of arsenic in irrigation water be 0.10 mg/l for continuous use on all soils and 2 mg/l for use up to 20 years on fine textured neutral to alkaline soils.

Beryllium

Haas (1932)[408] reported that some varieties of citrus seedlings showed toxicities at 2.5 mg/l of beryllium whereas others showed toxicity at 5 mg/l in nutrient solutions. Romney et al. (1962)[455] found that beryllium at 0.5 mg/l in nutrient solutions reduced the growth of bush beans. Romney and Childress (1965)[454] found that 2 mg/l or greater in nutrient solutions reduced the growth of tomatoes, peas, soybeans, lettuce, and alfalfa plants. Additions of soluble beryllium salts at levels equivalent to 4 per cent of the cation-adsorption capacity of two acid soils reduced the yields of ladino clover. Beryllium carbonate and beryllium oxide at the same levels did not reduce yields. These results suggest that beryllium in calcareous soils might be much less active and less toxic than in acid soils. Williams and LeRiche (1968)[480] found that beryllium at 2 mg/l in nutrient solutions was toxic to mustard, whereas 5 mg/l was required for growth reductions with kale.

It seems reasonable to recommend low levels of beryllium in view of the fact that, at 0.1 mg/l, 80 pounds of beryllium would be added in 100 years using 3 acre feet of water per acre per year. In 20 years, at 0.5 mg/l, water at the same rate would add 80 pounds.

Recommendations

In view of toxicities in nutrient solutions and in soils, it is recommended that maximum concentrations of beryllium in irrigation waters be 0.10 mg/l for continuous use on all soils and 0.50 mg/l for use on neutral to alkaline fine textured soils for a 20-year period.

Boron

Boron is an essential element for the growth of plants. Optimum yields of some plants are obtained at concentrations of a few tenths mg/l in nutrient solutions. However, at concentrations of 1 mg/l, boron is toxic to a number of sensitive plants. Eaton (1935,[400] 1944[401]) determined the boron tolerance of a large number of plants and developed lists of sensitive, semitolerant, and tolerant species. These lists, slightly modified, are also given in the U.S.D.A. Handbook 60 (Salinity Laboratory 1954)[459] and are presented in Table V-14. In general, sensitive crops showed toxicities at 1 mg/l or less, semitolerant crops at 1 to 2 mg/l, and tolerant crops at 2 to 4 mg/l. At concentrations above 4 mg/l, the irrigation water was generally unsatisfactory for most crops.

TABLE V-14—Relative Tolerance of Plants to Boron

(In each group the plants first named are considered as being more tolerant and the last named more sensitive.)

Tolerant	Semitolerant	Sensitive
Athel (Tamarix asphylla)	Sunflower (native)	Pecan
Asparagus	Potato	Black Walnut
Palm (Phoenix canariensis)	Acala cotton	Persian (English) walnut
Date palm (P. dactylifera)	Pima cotton	Jerusalem artichoke
Sugar beet	Tomato	Navy bean
Mangel	Sweetpea	American elm
Garden beet	Radish	Plum
Alfalfa	Field pea	Pear
Gladiolus	Ragged Robin rose	Apple
Broadbean	Olive	Grape (Sultanina and Malaga)
Onion	Barley	Kadota fig
Turnip	Wheat	Persimmon
Cabbage	Corn	Cherry
Lettuce	Milo	Peach
Carrot	Oat	Apricot
	Zinnia	Thornless blackberry
	Pumpkin	Orange
	Bell pepper	Avocado
	Sweet potato	Grapefruit
	Lima bean	Lemon

Salinity Laboratory Staff 1954[459].

Bradford (1966),[379] in a review of boron deficiencies and toxicities, stated that when the boron content of irrigation waters was greater than 0.75 mg/l, some sensitive plants, such as citrus, begin to show injury. Chapman (1968)[387] concluded that citrus showed some mild toxicity symptoms when irrigation waters have 0.5 to 1.0 mg/l, and that when the concentration was greater than 10 mg/l pronounced toxicities were found.

Biggar and Fireman (1960)[375] and Hatcher and Bower (1958)[411] showed that the accumulation of boron in soils is an adsorption process, and that before soluble levels of 1 or 2 mg/l can be found, the adsorptive capacity must be saturated. With neutral and alkaline soils of high adsorption capacities water of 2 mg/l might be used for some time without injury to sensitive plants.

Recommendations

From the extensive work on citrus, one of the most sensitive crops, the maximum concentration of 0.75 mg boron/l for use on sensitive crops on all soils seems justified. Recommended maximum concentrations for semitolerant and tolerant plants are considered to be 1 and 2 mg/l respectively.

For neutral and alkaline fine textured soils the recommended maximum concentration of boron in irrigation water used for a 20-year period on sensitive crops is 2.0 mg/l. With tolerant plants or for shorter periods of time higher boron concentrations are acceptable.

Cadmium

Data by Page et al. *in press* (1972)[444] showed that the yields of beans, beets, and turnips were reduced about 25 per cent by 0.10 mg cadmium/l in nutrient solutions; whereas cabbage and barley gave yield decreases of 20 to 50 per cent at 1.0 mg/l. Corn and lettuce were intermediate in response with less than 25 per cent yield reductions at 0.10 mg/l and greater than 50 per cent at 1.0 mg/l. Cadmium contents of plants grown in soils containing 0.11 to 0.56 mg/l acid extractable cadmium (Lagerwerff 1971)[427] were of the same order of magnitude as the plants grown by Page et al. in control nutrient solutions.

Because of the phytotoxicity of cadmium to plants, its accumulation in plants, lack of soils information, and the potential problems with this element in foods and feeds, a conservative approach is taken.

Recommendations

Maximum concentrations for cadmium in irrigation waters of 0.010 mg/l for continuous use on all soils and 0.050 mg/l on neutral and alkaline fine textured soils for a 20-year period are recommended.

Chromium

Even though a number of investigators have found small increases in yields with small additions of this element, it has not become recognized as an essential element. The primary concern of soil and plant scientists is with its toxicity. Soane and Saunders (1959)[466] found that 10 mg/l of chromium in sand cultures was toxic to corn, and that for tobacco 5 mg/l of chromium caused reduced growth and 1.0 mg/l reduced stem elongation. Scharrer and Schropp (1935)[461] found that chromium, as chromic sulfate, was toxic to corn at 5 mg/l in nutrient solutions. Hewitt (1953)[412] found that 8 mg/l chromium as chromic or chromate ions produced iron chlorosis on sugar beets grown in sand cultures. Hewitt also found that the chromate ion was more toxic than the chromic ion. Hunter and Vergnano (1953)[421] found that 5 mg/l of chromium in nutrient solutions produced iron deficiencies in plants. Turner and Rust (1971)[470] found that chromium treatments as low as 0.5 mg/l in water cultures and 10 mg/kg in soil cultures significantly reduced the yields of two varieties of soybeans.

Because little is known about the accumulation of chromium in soils in relation to its toxicity, a concentration of less than 1.0 mg/l in irrigation waters is desirable. At this concentration, using 3 acre feet water/acre/yr, more than 80 lb of chromium would be added per acre in 100 years, and using a concentration of 1.0 mg/l for a period of 20 years and applying water at the same rate, about 160 pounds of chromium would be added to the soil.

Recommendations

In view of the lack of knowledge concerning chromium accumulation and toxicity, a maximum concentration of 0.1 mg/l is recommended for continuous use on all soils and 1.0 mg/l on neutral and alkaline fine textured soils for a 20-year period is recommended.

Cobalt

Ahmed and Twyman (1953)[365] found that tomato plants showed toxicity from cobalt at 0.1 mg/l, and Vergnano and Hunter (1953)[475] found that cobalt at 5 mg/l was highly toxic to oats. Scharrer and Schropp (1933)[460] found that cobalt at a few mg/l in sand and solution cultures was toxic to peas, beans, oats, rye, wheat, barley, and corn, and that the tolerance to cobalt increased in the order listed. Vanselow (1966a)[473] found additions of 100 mg/kg to soils were not toxic to citrus.

The literature indicates that a concentration of 0.10 mg/l for cobalt is near the threshold toxicity level in nutrient solutions. Thus, a concentration of 0.05 mg/l appears to be satisfactory for continuous use on all soils. However, because the reaction of this element with soils is strong at neutral and alkaline pH values and it increases with time (Hodgson 1960),[416] a concentration of 5.0 mg/l might be tolerated by fine textured neutral and alkaline soils when it is added in small yearly increments.

Recommendations

Recommended maximum concentrations for cobalt are set at 0.050 mg/l for continuous use on all soils and 5.0 mg/l for neutral and alkaline fine textured soils for a 20-year period.

Copper

Copper concentrations of 0.1 to 1.0 mg/l in nutrient solutions have been found to be toxic to a large number of plants (Piper 1939,[447] Liebig et al. 1942,[432] Frolich et al. 1966,[403] Nollendorfs 1969,[442] Struckmeyer et al. 1969,[469] Seillac 1971[462]). Westgate (1952)[478] found copper toxicity in soils that had accumulated 800 lb/acre from the use of Bordeaux sprays. Field studies in sandy soils of Florida (Reuther and Smith 1954)[457] showed that toxicity to citrus resulted when copper levels reached 1.6 mg/meq of cation-exchange capacity per 100 g of dry soil.

The management procedures that reduce copper toxicity include liming the soil if it is acid, using ample phosphate fertilizer, and adding iron salts (Reuther and Labanauskas 1966).[456]

Toxicity levels in nutrient solutions and limited data on soils suggest a concentration of 0.20 mg/l for continuous use on all soils. This level used at a rate of 3 acre feet of water per year would add about 160 pounds of copper in 100 years, which is approaching the recorded levels of toxicity in acid sandy soils. A safety margin can be obtained by liming these soils. A concentration of copper at 5.0 mg/l applied in irrigation water at the rate of 3 acre feet of water per year for a 20-year period would add 800 pounds of copper in 20 years.

Recommendations

Based on toxicity levels in nutrient solutions and the limited soils data available, a maximum concentration of 0.20 mg/l copper is recommended for continuous use on all soils. On neutral and alkaline fine textured soils for use over a 20-year period, a maximum concentration of 5.0 mg/l is recommended.

Fluoride

Applications of soluble fluoride salts to acid soils can produce toxicity to plants. Prince et al. (1949)[450] found that 360 pounds fluoride per acre, added as sodium fluoride, reduced the yields of buckwheat at pH 4.5, but at pH values above 5.5 this rate produced no injury.

MacIntire et al. (1942)[435] found that 1,150 pounds of fluoride in superphosphate, 575 pounds of fluoride in slag, or 2,300 pounds of fluoride as calcium fluoride per acre had no detrimental effects on germination or plant growth on well-limed neutral soils, and that vegetation grown on these soils showed only a slight increase in fluoride as compared to those grown in acid soils. However, Shirley et al. (1970)[461] found that bones of cows that had grazed pastures fertilized with raw rock and colloidal phosphate, which contained approximately two to three per cent fluorides, for seven to 16 years averaged approximately 2,900 and 2,300 mg of fluorine per kilogram of bone, respectively. The bones of cows that had grazed on pastures fertilized with relatively fluorine free superphosphate, concentrated superphosphate, and basic slag fertilizer contained only 1400 mg/kg fluorine.

Recommendations

Because of the capacity of neutral and alkaline soils to inactivate fluoride, a relatively high maximum concentration for continuous use on these soils is recommended. Recommended maximum concentrations are 1.0 mg/l for continuous use on all soils and 15 mg/l for use for a 20-year period on neutral and alkaline fine textured soils.

Iron

Iron in irrigation waters is not likely to create a problem of plant toxicities. It is so insoluble in aerated soils at all pH values in which plants grow well, that it is not toxic. In fact, the problems with this element are deficiencies in alkaline soils. In reduced (flooded) soils soluble ferrous ions develop from inherent compounds in soils, so that quantities that might be added in waters would be of no concern. However, Rhoads (1971)[458] found large reductions in the quality of cigar wrapper tobacco when plants were sprinkler irrigated with water containing 5 or more mg soluble iron/l, because of precipitation of iron oxides on the leaves. Rhoad's experience would suggest caution when irrigating any crops using sprinkler systems and waters having sufficient reducing conditions to produce reduced and soluble ferrous iron.

The disadvantages of soluble iron salts in waters are that these would contribute to soil acidification, and the precipitated iron would increase the fixation of such essential elements as phosphorous and molybdenum.

Recommendations

A maximum concentration of 5.0 mg/l is recommended for continuous use on all soils, and a maximum concentration of 20 mg/l is recommended on neutral to alkaline soils for a 20-year period. The use of waters with large concentrations of suspended freshly precipitated iron oxides and hydroxides is not recommended, because these materials also increase the fixation of phosphorous and molybdenum.

Lead

The phytotoxicity of lead is relatively low. Berry (1924)[374] found that a concentration of lead nitrate of 25 mg/l was required for toxicity to oats and tomato plants. At a concentration of 50 mg/l, death of plants occurred. Hopper (1937)[418] found that 30 mg/l of lead in nutrient solutions was toxic to bean plants. Wilkins (1957)[479] found that lead at 10 mg/l as lead nitrate reduced root growth. Since soluble lead contents in soils were usually from 0.05 to 5.0 mg/kg (Brewer 1966),[383] little toxicity can be expected. It was shown that the principal entry of lead into plants was from aerial deposits rather than from absorption from soils (Page et al. 1971)[445] indicating that lead that falls onto the soil is not available to plants.

In a summary on the effects of lead on plants, the Committee on the Biological Effects of Atmosphere Pollutants (NRC 1972)[441] concluded that there is not sufficient evidence to indicate that lead, as it occurs in nature, is toxic to vegetation. However, in studies using roots of some plants and very high concentrations of lead, this element was reported to be concentrated in cell walls and nuclei during mitosis and to inhibit cell proliferation.

Recommendations

Recommended maximum concentrations of lead are 5.0 mg/l for continuous use on all soils and 10 mg/l for a 20-year period on neutral and alkaline fine textured soils.

Lithium

Most crops can tolerate lithium in nutrient solutions at concentrations up to 5 mg/l (Oertli 1962,[443] Bingham et al. 1964,[377] Bollard and Butler 1966[37]). But research revealed that citrus was more sensitive (Aldrich et al. 1951,[369] Bradford 1963b,[381] Hilgeman et al. 1970[415]). Hilgeman et al. (1970)[415] found that grapefruit developed severe symptoms of lithium toxicity when irrigated with waters containing lithium at 0.18 to 0.25 mg/l. Bradford (1963a)[380] reported that experience in California indicated slight toxicity of lithium to citrus at 0.06 to 0.10 mg/l in the water.

Lithium is one of the most mobile of cations in soils. It tends to be replaced by other cations in waters or fertilizers and is removed by leaching. On the other hand, it is not precipitated by any known process.

Recommendations

Recommendations for maximum concentrations of lithium, based on its phytotoxicity, are 2.5 mg/l for continuous use on all soils, except for citrus where the recommended maximum concentration is 0.075 mg/l for all soils. For short-term use on fine textured soils the same maximum concentrations are recommended because of lack of inactivation in soils.

Manganese

Manganese concentrations at a few tenths to a few milligrams per liter in nutrient solutions are toxic to a number of crops as shown by Morris and Pierre (1949),[440] Adams and Wear (1957),[364] Hewitt (1965),[414] and others. However, toxicities of this element are associated with acid soils, and applications of proper quantities of ground limestone successfully eliminated the problem. Increasing the pH to the 5.5 to 6.0 range usually reduced the active manganese to below the toxic level (Adams and Wear 1957).[364] Hoyt and Nyborg (1971b)[420] found that available manganese in the soil and manganese content of plants were negatively correlated with soil pH. However, the definite association of toxicity with soil pH as found with aluminum was not found with manganese, which has a more complex chemistry. Thus, more care must be taken in setting water quality criteria for manganese than for aluminum (i.e., management for control of toxicities is not certain).

Recommendations

Recommended maximum concentrations for manganese in irrigation waters are set at 0.20 mg/l for continued use on all soils and 10 mg/l for use up to 20 years on neutral and alkaline fine textured soils. Concentrations for continued use can be increased with alkaline or calcareous soils, and also with crops that have higher tolerance levels.

Molybdenum

This element presents no problems of toxicity to plants at concentrations usually found in soils and waters. The problem is one of toxicity to animals from molybdenum ingested from forage that has been grown in soils with relatively high amounts of avaiable molybdenum. Dye and O'Hara (1959)[398] reported that the molybdenum concentration in forage that produced toxicity in ruminants was 5 to 30 mg/kg. Lesperance and Bohman (1963)[430] found that toxicity was not simply associated with the molybdenum content of forage but was influenced by the amounts of other elements, particularly copper. Jensen and Lesperance (1971)[423] found that the accumulation of molybdenum in plants was proportional to the amount of the element added to the soil.

Kubota et al. (1963)[426] found that molybdenum concentrations of 0.01 mg/l or greater in soil solutions were associated with animal toxicity levels of this element in alsike clover. Bingham et al. (1970)[376] reported that molybdosis of cattle was associated with soils that had 0.01 to 0.10 mg/l of molybdenum in saturation extracts of soils.

Recommendations

The recommended maximum concentration of molybdenum for continued use of water on all soils, based on animal toxicities from forage, is 0.010 mg/l. For short term use on soils that react with this element, a concentration of 0.050 mg/l is recommended.

Nickel

According to Vanselow (1966b),[474] many experiments with sand and solution cultures have shown that nickel at 0.5 to 1.0 mg/l is toxic to a number of plants. Chang and Sherman (1953)[385] found that tomato seedlings were injured by 0.5 mg/l. Millikan (1949)[437] found that 0.5 to 5.0 mg/l were toxic to flax. Brenchley (1938)[382] reported toxicity to barley and beans from 2 mg/l. Crooke (1954)[395] found that 2.5 mg/l was toxic to oats. Legg and Ormerod (1958)[429] found that 1.0 mg/l produced toxicity in hop plants. Vergnano and Hunter (1953)[475] found that 1.0 mg/l in solutions flushed through sand cultures was toxic to oats. Soane and Saunders (1959)[466] found that tobacco plants showed no toxicity at 30 mg/l, and that corn showed no toxicity at 2 mg/l but showed toxicity at 10 mg/l.

Work by Mizuno (1968)[439] and Halstead et al. (1969)[409] and the review of Vanselow (1966b)[474] showed that increasing the pH of soils reduces the toxicity of added nickel.

Halstead et al. (1969)[409] found the greatest capacity to adsorb nickel without development of toxicity was by a soil with 21 per cent organic matter.

Recommendations

Based on both toxicity in nutrient solutions and on quantities that produce toxicities in soils, the recommended maximum concentration of nickel in irrigation waters is 0.20 mg/l for continued use on all soils. For neutral fine textured soils for a period up to 20 years, the recommended maximum is 2.0 mg/l.

Selenium

Selenium is toxic at low concentrations in nutrient solutions, and only small amounts added to soils increase the selenium content of forages to a level toxic to livestock. Broyer et al. (1966)[384] found that selenium at 0.025 mg/l in nutrient solutions decreased the yields of alfalfa.

The best evidence for use in setting water quality criteria for this element is application rates in relation to toxicity in forages. Amounts of selenium in forages required to prevent selenium deficiencies in cattle (Allaway et al. 1967)[366] ranged between 0.03 and 0.10 mg/kg (depending on other factors), whereas concentrations above 3 or 4 mg/kg were considered toxic (Underwood 1966).[471] A number of investigators (Hamilton and Beath 1963,[410] Grant 1965,[407] Allaway et al. 1966)[367] have shown that small applications of selenium to soils at a rate of a few kilograms per hectare produced plant concentrations of selenium that were toxic to animals. Gissel-Nielson and Bisbjerg (1970)[406] found that applications of approximately 0.2 kg/hectare of selenium produced from 1.0 to 10.5 mg/kg in tissues of forage and vegetable crops.

Recommendation

With the low levels of selenium required to produce toxic levels in forages, the recommended maximum concentration in irrigation waters is 0.02 mg/l for continuous use on all soils. At a rate of 3 acre feet of water per acre per year this concentration represents 3.2 pounds per acre in 20 years. The same recommended maximum concentration should be used on neutral and alkaline fine textured soils until greater information is obtained on soil reactions. The relative mobility of this element in soils in comparison to other trace elements and slow removal in harvested crops provide a sufficient safety margin.

Tin, Tungsten, and Titanium

Tin, tungsten, and titantium are effectively excluded by plants. The first two can undoubtedly be introduced to plants under conditions that can produce specific toxicities. However, not enough is known at this time about any of the three to prescribe tolerance limits. (This is true with other trace elements such as silver.) Titantium is very insoluble, at present it is not of great concern.

Vanadium

Gericke and Rennenkampff (1939)[405] found that vanadium at 0.1, 1.0, and 2.0 mg/l added to nutrient solutions as calcium vanadate slightly increased the growth of barley, whereas at 10 mg/l vanadium was toxic to both tops and roots and that vanadium chloride at 1.0 mg/l of vanadium was toxic. Warington (1954,[476] 1956[477]) found that flax, soybeans, and peas showed toxicity to vanadium in the concentration range of 0.5 to 2.5 mg/l. Chiu (1953)[389] found that 560 pounds per acre of vanadium added as ammonium metavanadate to rice paddy soils produced toxicity to rice.

Recommendations

Considering the toxicity of vanadium in nutrient solutions and in soils and the lack of information on the reaction of this element with soils, a maximum concentration of 0.10 mg/l for continued use on all soils is recommended. For a 20-year period on neutral and alkaline fine textured the recommended maximum concentration is 1.0 mg/l.

Zinc

Toxicities of zinc in nutrient solutions have been demonstrated for a number of plants. Hewitt (1948)[413] found that zinc at 16 to 32 mg/l produced iron deficiencies in sugar beets. Hunter and Vergnano (1953)[421] found toxicity to oats at 25 mg/l. Millikan (1947)[438] found that 2.5 mg/l produced iron deficiency in oats. Earley (1943)[399] found that the Peking variety of soybeans was killed at 0.4 mg/l, whereas the Manchu variety was killed at 1.6 mg/l.

The toxicity of zinc in soils is related to soil pH, and liming acid soil has a large effect in reducing toxicity (Barnette 1936,[371] Gall and Barnette 1940,[404] Peech 1941,[446] Staker and Cummings 1941,[468] Staker 1942,[467] Lee and Page 1967[428]). Amounts of added zinc that produce toxicity are highest in clay and peat soils and smallest in sands.

On acid sandy soils the amounts required for toxicity would suggest a recommended maximum concentration of zinc of 1 mg/l for continuous use. This concentration at a water application rate of 3 acre feet/acre/year would add 813 pounds per acre in 100 years. However, if acid sandy soils are limed to pH values of six or above, the tolerance level is increased by at least a factor of two (Gall and Barnette 1940).[404]

Recommendations

Assuming adequate use of liming materials to keep pH values high (six or above), the recommended maximum concentration for continuous use on all soils is 2.0 mg/l. For a 20-year period on neutral and alkaline soils the recommended maximum is 10 mg/l. On fine textured calcareous soils and on organic soils, the concentrations can exceed this limit by a factor of two or three with low probability of toxicities in a 20-year period.

PESTICIDES (IN WATER FOR IRRIGATION)

Pesticies are used widely in water for irrigation on commercial crops in the United States (Sheets 1967).[502] Figures on production, acreage treated, and use patterns indicate insecticides and herbicides comprise the major agricultural pesticides. There are over 320 insecticides and 127 herbicides registered for agricultural use (Fowler 1972).[498]

Along with the many benefits to agriculture, pesticides can have detrimental effects. Of concern for irrigated agriculture is the possible effects of pesticide residues in irrigation water on the growth and market quality of forages and crops. Pesticides most likely to be found in agricultural water supplies are listed in the Freshwater Appendix II-D.

Insecticides in Irrigation Water

The route of entry of insecticides into waters is discussed in the pesticide section under Water for Livestock Enterprises. For example, Miller et al. (1967)[500] observed the movement of parathion from treated cranberry bogs into a nearby irrigation ditch and drainage canal, and Sparr et al. (1966)[503] monitored endrin in waste irrigation water used three days after spraying. In monitoring pesticides in water used to irrigate areas near Tule Lake and lower Klamath Lake Wildlife Refuges in northern California, Godsil and Johnson (1968)[illegible] detected high levels of endrin compared to other pesticides. They observed that the concentrations of pesticides in irrigation waters varied directly with agricultural activities.

In monitoring pesticides residues from 1965 to 1967 (Agricultural Research Service 1969a),[183] the U. S. Department of Agriculture detected the following pesticides in irrigation waters at a sampling area near Yuma, Arizona: the DDT complex, dieldrin, methyl parathion, endrin, endosulfan, ethyl parathion, dicofol, s,s,s,-tributyl phosphorotrithiate (DEF), and demeton. Insecticides most commonly detected were DDT, endrin, and dieldrin. For the most part, all residues in water were less than 1.0 μg/l. A further examination of the irrigation water at the Yuma sampling area showed that water entering it contained relatively low amounts of insecticide residues while water leaving contained greater concentrations. It was concluded that some insecticides were picked up from the soil by irrigation water and carried out of the fields.

Crops at the same location were also sampled for insecticide residues. With the exception of somewhat higher concentrations of DDT and dicofol in cotton stalks and cantaloupe vines, respectively, residues in crop plants were relatively small. The mean concentrations, where detected, were 2.6 μg/g combined DDT, 0.01 μg/g endrin, 0.40 μg/g dieldrin, 0.05 μg/g lindane, 5.0 μg/g dicofol, and 1.8 μg/g combined parathion. The larger residues for DDT and dicofol were apparently from foliage applications. Sampling of harvested crops showed that residues were generally less than 0.30 μg/g and occurred primarily in lettuce and in cantaloupe pulp, seeds, and rind. DDT, dicofol, and endrin were applied to crops during the survey, and from 2.0 to 6.0 lb/acre of DDT were applied to the soil before 1965.

Some crops do not absorb measurable amounts of insecticides but others translocate the chemicals in various amounts. At the levels (less than 1.0 μg/l) monitored by the U. S. Department of Agriculture in irrigation waters (Agricultural Research Service 1969a),[183] there is little evidence indicating that insecticide residues in the water are detrimental to plant growth or accumulate to undesirable or illegal concentrations in food or feed crops.

Herbicides in Irrigation Water

In contrast to insecticides, misuse of herbicides can present a greater hazard to crop growth. Herbicides are likely to be found in irrigation water under the following circumstances: (1) during their purposeful introduction into irrigation water to control submersed weeds; of (2) incidental to herbicide treatment for control of weeds on banks of irrigation canals. Attempts are seldom made to prevent water containing herbicides such as xylene or acrolein from being diverted onto cropland during irrigation. In most instances, however, water-use restrictions do apply when herbicides are used in reservoirs of irrigation water. The herbicides used in reservoirs are more persistent and inherently more phytotoxic at low levels than are xylene and acrolein.

The tolerances of a number of crops to various herbicides used in and around water are listed in Table V-15. Residue levels tolerated by most crops are usually much higher than the concentrations found in water following normal use of the herbicides. Aromatic solvent (xylene) and acrolein are widely used in western states for keeping irrigation canals free of submersed weeds and algae and are not harmful to crops at concentrations needed for weed control. (U. S. Department of Agriculture, Agricultural Research Service 1963,[201] hereafter referred to as Agricultural Research Service 1963).[182] Xylene, which is non-polar, is lost rapidly from water (50 per cent in 3 to 4 hours) by volatility (Frank et al. 1970).[197] Acrolein, a polar compound, may remain in flowing water for periods of 24 hours or more at levels that are phytotoxic only to submersed aquatic weeds. Copper sulfate is used frequently to control algae. It has also been found effective on submersed vascular weeds when applied continuously to irrigation water at low levels (Bartley 1969).[187]

The herbicides that have been used most widely on irrigation ditchbanks are 2,4-D, dalapon, TCA, and silvex. The application of herbicides may be restricted to a swath of a few feet along the margin of the water, or it may cover a swath 15 feet or more wide. A variable overlap of the spray pattern at the water margin is unavoidable and accounts for most of the herbicide residues that occur in water during ditchbank treatments. Rates of application vary from 2 lb per acre for 2,4-D to 20 lb per acre for dalapon. For examples of residue levels that occur in water from these treatments see Table V-16. The residues generally occur only during the periods when ditchbanks are treated.

The rates of dissipation of herbicides in irrigation water were reported recently by Frank et al. (1970).[197] The herbicides and formulations commonly used on ditchbanks are readily soluble in water and not extensively sorbed to soil or other surfaces. Reduction in levels of residues in flowing irrigation water is due largely to dilution. Irrigation canals

TABLE V-15—Tolerance of Crops to Various Herbicides Used In and Around Waters[a]

Herbicide	Site of use	Formulation	Treatment rate	Concentration that may occur in irrigation water[b]	Crop injury threshold in irrigation water (mg/l)[c]
Acrolein	Irrigation canals	Liquid	15 mg/l for 4 hours	10 to 0.1 mg/l	Flood or furrow; beans-60, corn-60, cotton-80, soybeans-20, sugar beets-60.
			0.6 mg/l for 8 hours	0.4 to 0.02 mg/l	Sprinkler; corn-60, soybeans-15, sugar beets-15.
			0.1 mg/l for 48 hours	0.05 to 0.1 mg/l	
Aromatic solvents (xylene)	Flowing water in canals or drains	Emulsifiable liquid	5 to 10 gal/cfs (350 to 750 mg/l) applied in 30–60 minutes	700 mg/l or less	Alfalfa >1,600, beans-1,200, carrots-1,600, corn-3,000 cotton-1,600, grain sorghum >800, oats-2,400, potatoes-1,300, wheat >1,200.
Copper sulfate	Canals or reservoirs	Pentahydrate crystals	Continuous treatment 0.5 to 3.0 mg/l, slug treatment [illegible] to 1 lb (0.15 to 0.45 kg) per cfs water flow	0.04 to 0.8 mg/l during first 10 miles, 0.08 to 9.0 mg/l during first 10 to 20 miles.	Threshold is above these levels.
Dalapon	Banks of canals and ditches	Water soluble salt	15 to 30 lb/A or 17 to 34 kg/ha	Less than 0.2 mg/l	Beets >7.0, corn >0.35
Diquat	Injected into water or sprayed over surface	Liquid	3 to 5 mg/l, 1 to 1.5 lbs/A, or 1.2 to 1.7 kg/ha	Usually less than 0.1 mg/l	Beans-5.0, corn-125
Diuron	Banks and bottoms of small dry powder ditches	Wettable powder	Up to 64 lb/A or 72 kg/ha	No data	No data
Dichlobenil	Bottoms of dry canals	Granules or wettable powder	7 to 10 lb/A or 7.9 to 12.6 kg/ha	No data	Alfalfa-10, corn >10, soybeans-1.0, sugar beets-1.0 to 10.
Endothall	Ponds and reservoirs	Water soluble Na or K salts	1 to 4 mg/l	Absent or only traces	Corn-25, field beans-1.0, Alfalfa >10.0
Endothall amine salts	Reservoirs and static-water canals	Liquid or granules	0.5 to 2.5 mg/l	Absent or only traces	Corn >25, soybeans >25, sugar beets-25
Fenac	Bottoms of dry canals	Liquid or granules	10 to 20 lb/A or 12.6 to 25.2 kg/ha	Absent or only traces	Alfalfa-1.0, corn-10, soybeans-0.1, sugar beets-0.1 to 10.
Monuron	Banks and bottoms of small dry powder ditches	Wettable powder	Up to 64 lb/A or 72 kg/ha	No data	No data
Silvex	Woody plants and brambles on floodways, along canal, stream, or reservoir banks	Esters in liquid form	2 to 4 lb/A or 2.2 to 4.4 kg/ha	No data. Probably well under 0.1 mg/l	Corn >5.0, sugar beets and soybeans >0.02.
	Floating and emersed weeds in southern waterways	″	2 to 8 lb/A or 2.2 to 8.8 kg/ha	0.01 to 1.6 mg/l 1 day after application	″
TCA	Banks of canals and ditches	Water soluble salt	Up to 64 lb/A or 72 kg/ha	Usually less than 0.1 mg/l	No injury observed at levels used.
2,4-D amine	On banks of canals and ditches	Liquid	1 to 4 lb/A or 1.1 to 4.4 kg/ha	0.01 to 0.10 mg/l	Field beans >1.0, grapes-0.7, sugar beets >0.2, soybeans >0.02, corn-10, cucumbers, potatoes, sorghum, alfalfa, peppers >1.0.
	Floating and emersed weeds in southern canals and ditches	″	2 to 4 lb/A or 2.2 to 4.4 kg/ha	No data. Probably less than 0.1 mg/l	″
Picloram	For control of brush on watersheds	Liquids or granules	1 to 3 lb/A or 1.1 to 3.3 kg/ha	No data	Corn >10, field beans 0.1, sugar beets >1.0

[a] Sources of data included in this table are: U.S. Department of Agriculture, Agricultural Research Service (1969)[illegible], Arle and McRae (1959,[illegible] 1960[illegible]), Bruns (1954,[illegible] 1957,[illegible] 1964,[illegible] 1969[illegible]), Bruns and Clore (1958),[illegible] Bruns and Dawson (1959),[illegible] Bruns et al. (1955,[illegible] 1964,[illegible] unpublished data 1971[illegible]) Frank et al. (1970),[illegible] Yeo (1959)[illegible].

[b] Herbicide concentrations given in this column are the highest concentrations that have been found in irrigation water, but these levels seldom remain in the water when it reaches the crop.

[c] Unless indicated otherwise, all crop tolerance data were obtained by flood or furrow irrigation. Threshold of injury is the lowest concentration causing temporary or permanent injury to crop plants even though, in many instances, neither crop yield nor quality was affected.

are designed to deliver a certain volume of water to be used on a specific area of cropland. Water is diverted from the canals at regular intervals, and this systematically reduces the volume of flow. Consequently, little or no water remains at the ends of most canals where disposal of water containing herbicides might be troublesome.

Residues in Crops

Successful application of herbicides for control of algae and submersed vascular weeds in irrigation channels is dependent upon a continuous flow of water. Because it is impractical to interrupt the flow and use of water during the application of herbicides in canals or on canal banks, the herbicide-bearing water is usually diverted onto croplands. Under these circumstances, measurable levels of certain herbicides may occur in crops.

Copper sulfate is used most frequently for control of algae at concentrations that are often less than the suggested tolerance for this herbicide in potable water. Application rates may range from one third pound of copper sulfate per cubic-feet-second (cfs) of water flow to two pounds per cfs of water flow (Agriculture Research Service 1963).[482] Xylene is a common formulating ingredient for many pesticides and as such is often applied directly to crop plants. The distribution by furrow or sprinkler of irrigation water containing acrolein contributes to the rapid loss of this herbicide. Copper sulfate, xylene, and acrolein are of minor importance as sources of objectionable residues in crops.

Phenoxy herbicides, dalapon, TCA, and amitrole are most persistent in irrigation water (Bartley and Hattrup 1970).[488] It is possible to calculate the maximum amount of a herbicide such as 2,4-D that might be applied to crop-

TABLE V-16—Maximum Levels of Herbicide Residues Found in Irrigation Water as a Result of Ditchbank Treatment[a]

Herbicide and canal treated	Treatment rate, lb/A	Water flow in cfs	Maximum concentration of residue, µg/l
DALAPON			
Five-mile Lateral	20	15	365[b]
Lateral No. 4	6.7	290	23
Manard Lateral	9.6	37	39
Yolo Lateral	10.5	26	162
TCA			
Lateral No. 4	3.8	290	12
Manard Lateral	5.4	37	20
Yolo Lateral	5.9	26	89
2.4-D AMINE SALT			
Lateral No. 4	1.9	290	5
Manard Lateral	2.7	37	13
Yolo Lateral	3.0	26	36

[a] Frank et al. (1970)[497].

[b] High level of residue probably due to atypical treatment.

land following its use on an irrigation bank. A four-mile-long body of irrigation water contaminated with 2,4-D and flowing at a velocity of one mile per hour, would be diverted onto an adjacent field for a period of 4 hours. A diversion rate of two acre inches of water in 10 hours would deliver 0.8 inch of contaminated water per acre. If this amount of water contained 50 µg/l of 2,4-D (a higher concentration than is usually observed), it would deposit slightly less than 0.009 lb of 2,4-D per acre of cropland. Levels of 2,4-D residues of greater magnitude have not caused injury to irrigated crops (see Table V-15).

The manner in which irrigation water containing herbicides is applied to croplands may influence the presence and amounts of residues in crops (Stanford Research Institute 1970).[509] For example, residues in leafy crops may be greater when sprinkler irrigated than when furrow irrigated, and the converse may be true with root crops.

If there is accidental contamination of field, forage, or vegetable crops by polluted irrigation water, the time interval between exposure and harvesting of the crop is important, especially with crops used for human consumption. Factors to be considered with those mentioned above include the intensity of the application, stage of growth, dilution, and pesticide degradability in order to assess the amount of pesticide that may reach the ultimate consumer (U. S. Department of Health, Education and Welfare 1969).[506] Pesticides applied to growing plants may affect the market quality by causing changes in the chemical composition, appearance, texture, and flavor of the product harvested for human consumption (NRC 1968).[501]

Recommendation

Pesticide residues in irrigation waters are variable depending upon land and crop management practices. Recent data indicate pesticide residues are declining in irrigation waters, with concentrations less than 1.0 µg/l being detected. To date there have been no documented toxic effects on crops irrigated with waters containing insecticide residues. Because of these factors and the marked variability in crop sensitivity, no recommendation is given for insecticide residues in irrigation waters. For selected herbicides in irrigation water, it is recommended that levels at the crop not exceed the recommended maximum concentration listed in Table V-16.

PATHOGENS

Plant Pathogens

The availability of "high quality" irrigation water may lead to the reuse of runoff water or tailwater and subsequently lead to a serious but generally unrecognized problem, that of the distribution of plant pathogenic organisms such as bacteria, fungi, nematodes, and possibly viruses. This is most serious when it occurs on previously nonfarmed lands.

Distribution of Nematodes Wide distribution of plant-nematodes in irrigation waters of south central Washington and the Columbia Basin of eastern Washington was demonstrated by Faulkner and Bolander (1966,[515] 1970[516]). When surface drainage from agricultural fields is collected and reintroduced into irrigation systems, without first being impounded in settling basins, large numbers of nematodes can be transferred. Faulkner and Bolander's data indicated that an acre of land in the Lower Yakima Valley may receive from 4 million to over 10 million plant-parasitic nematodes with each irrigation. Numbers of nematodes transported vary with the growing season, but some that were detectable in irrigation water and demonstrated to be infective were *Meloidogyne hapla*, *Heterodera schachtii*, *Pratylenchus* sp., and *Tylenchorhynchus* sp.

Meagher (1967)[526] found that plant-parasitic nematodes such as the citrus nematode, *Tylenchulus semipenetrans*, may be spread by subsoil drainage water reused for irrigation.

Thomason and Van Gundy (1961)[530] showed another means by which nematodes may possibly enter irrigation supplies. Two species of rootknot nematode, *Meloidogyne incognita* and *M. javanica*, were found reproducing on arrowweed, *Pluchea sericea*, at the edge of sandbars in the Colorado River at Blythe, California. No conclusive evidence that nematodes entered the river was presented, but infested soil and infected roots were in direct contact with the water.

Plant-parasitic nematodes are essentially aquatic animals and may survive for days or weeks immersed in water. Unless provisions are made for excluding them from or settling them out of irrigation water, they may seriously deteriorate water quality in areas of the United States dependent on irrigation for crop production.

Distribution of Fungi Surveys were conducted to determine the origins and prevalence of *Phytophthora* sp., a fungus pathogenic to citrus, in open irrigation canals and reservoirs in five southern California counties by Klotz et al. (1959).[523] *Phytophthora* progagules were detected by trapping them on healthy lemon fruits suspended in the water.

Of the 12 canals tested from September 1957 to September 1958, all yielded *Phytophthora* sp. at one time or another, some more consistently than others. *Phytophthora citrophthora* was the most common and was recovered from 11 canals.

In the five canals where it was possible to set the lemon traps at the source of the water, no *Phytophthora* sp. were recovered. However, as the canals passed through citrus areas where excess irrigation water or rain runoff could drain into the canals, the fungi were readily isolated. Soil samples collected from paths of runoff water that drained into irrigation canals yielded *P. citrophthora*, indicating that *Phytophthora* zoospores from infested citrus groves can be introduced into canals.

One of three reservoirs was found to be infested with *P. parasitica*. Application of copper sulfate effectively controlled the fungus under the static condition of the water in the reservoir. Chlorination (2 mg/l for 2 minutes) effectively killed the infective zoospores of *Phytophthora* sp. under laboratory conditions.

McIntosh (1966)[525] established that *Phytophthora cactorum*, which causes collar-rot of fruit trees in British Columbia, contaminates the water of many irrigation systems in the Okanagan and Similkamen Valleys. The fungus was isolated from 15 sources including ponds, reservoirs, rivers, creeks, and canals. It had been established previously that *P. cactorum* was widespread in irrigated orchard soils of the area, but could not be readily detected in nonirrigated soils.

Many plant-pathogenic fungi normally produce fruiting bodies that are widely disseminated by wind. A number do not, however, and these could easily be disseminated by irrigation water.

Distribution of Viruses Most plant pathogenic viruses do not remain infestive in the soil outside the host or vector. Two exceptions may be tobacco mosaic virus (TMV) and tobacco necrosis virus (TNV). There is some evidence that these persist in association with soil colloids and can gain entry to plant roots through wounds. Hewitt et al. (1958)[520] demonstrated that fan leaf virus of grape is transmitted by a dagger nematode, *Xiphinema index*. To date, three genera of nematodes, *Xiphinema*, *Longidorus*, and *Trichodorus* are known to transmit viruses. The first two of these genera transmit polyhedral viruses of the Arabis mosaic group. *Trichdorus* spp. transmit tubular viruses of the Tobacco Rattles group.

Infective viruses are known to persist in the nematode vector for months in the absence of a host plant. This information, coupled with Faulkner and Bolander's (1966,[515] 1970)[516] proof of the distribution of nematodes in irrigation water, suggested the possibility that certain plant viruses could be distributed in their nematode vectors in irrigation water. To date, no direct evidence for this has been published.

Several other soil-borne plant-pathogenic viruses are transmitted to hosts by soil fungi. The ability of the fungus *Olpidium brassicae* to carry and transmit Lettuce Big Vein Virus (LBVV) was recently demonstrated (Grogan et al. 1958,[519] Campbell 1962,[513] Teakle 1969[529]). It is carried within the zoospore into fresh roots and there released. The most likely vehicle for its distribution in irrigation water would be resting sporangia carried in runoff water from infested fields. The resting sporangia are released into the soil from decaying roots of host plants. Another economically important virus transmitted by a soil fungus is Wheat Mosaic Virus carried by the fungus *Polymyxa graminis* (Teakle 1969).[529]

Another means of spread of plant viruses (such as Tobacco Rattles Virus and Arabis Mosaic Viruses that are vectored by nematodes) is through virus-infected weed seed carried in irrigation water.

Distribution of Bacteria Bacterial plant pathogens would appear to be easily transported in irrigation water. However, relatively few data have been published concerning these pathogens. Kelman (1953)[522] reported the spread of the bacterial wilt organism of tobacco in drainage water from fields and in water from shallow wells. He also noted spread of the disease along an irrigation canal carrying water from a forested area, but no direct evidence of the bacterium in the water was presented. Local spread in runoff water is substantiated but not in major irrigation systems.

Controlling plant disease organisms in irrigation water should be preventive rather than an attempt to remove them once they are introduced. In assuring that irrigation water does not serve for the dispersal of important plant pathogens, efforts should be directed to those organisms that are not readily disseminated by wind, insects, or other means. Attention should be focused on those soil-borne nematodes, fungi, viruses, and bacteria that do not spread rapidly in nature.

Two major means of introduction of plant pathogens into irrigation systems are apparent. The most common is natural runoff from infested fields and orchards during heavy rainfall and floods. The other is collection of irrigation runoff or tailwater and its return to irrigation canals. If it is necessary to trap surface water, either from rainfall or irrigation drainage, provisions should be made to impound the water for sufficient time to allow settling out of nematodes and possibly other organisms.

Water may be assayed for plant pathogens, but there are thousands, or perhaps millions of harmless microorganisms for every one that causes a plant disease. However, plant pathogenic nematodes, and perhaps certain fungi, can be readily trapped from irrigation water, easily identified, and used as indicators of contamination (Klotz et al. 1959,[523] Faulkner and Bolander 1966,[515] McIntosh 1966[525]).

Plant infection is not considered serious unless an economically important percentage of the crop is affected. The real danger is that a trace of plant disease can be spread by water to an uninfected area, where it can then be spread by other means and become important. It is unlikely that any method of water examination would be as effective in preventing this as would the prohibitions such as those suggested above.

Human and Animal Pathogens

Many microorganisms, pathogenic for either animals or humans, or both, may be carried in irrigation water, particularly that derived from surface sources. The list comprises a large variety of bacteria, spirochetes, protozoa, helminths, and viruses which find their way into irrigation water from municipal and industrial wastes, including food-processing plants, slaughterhouses, poultry-processing operations, and feedlots. The diseases associated with these organisms include bacillary and amebic dysentery, *Salmonella* gastroenteritis, typhoid and paratyphoid fevers, leptospirosis, cholera, vibriosis, and infectious hepatitis. Other less common infections are tuberculosis, brucellosis, listeriosis, coccidiosis, swine erysipelas, ascariasis, cysticercosis and tapeworm disease, fascioliasis, and schistosomiasis.

Of the types of irrigation commonly practiced, sprinkling requires the best quality of water from a microbiological point of a view, as the water and organisms are frequently applied directly to that portion of the plant above the ground, especially fruits and leafy crops such as strawberries, lettuce, cabbage, alfalfa, and clover which may be consumed raw by humans or animals. Flooding the field may pose the same microbiological problems if the crop is eaten without thorough cooking. Subirrigation and furrow irrigation present fewer problems as the water rarely reaches the upper portions of the plant; and root crops, as well as normal leafy crops and fruits, ordinarily do not permit penetration of the plant by animal and human pathogens. Criteria for these latter types may also depend upon the characteristics of the soil, climate and other variables which affect survival of the microorganisms.

Benefits can be obtained by coordinating operation of reservoir releases with downstream inflows to provide sedimentation and dilution factors to markedly reduce the concentrations of pathogens in irrigation water (LeBosquet 1945,[524] Camp et al. 1949[512]).

The common liver fluke, *Fasciola hepatica*, the ova of which are spread from the feces of many animals, commonly affects cattle and sheep (Allison 1930,[510] U.S. Dept. Agriculture 1961[531]), and may affect man. The intermediate hosts, certain species of snails, live in springs, slow-moving swampy waters, and on the banks of ponds, streams, and irrigation ditches. After development in the snail, the cercarial forms emerge and encyst on grasses, plants, bark, or soil. Cattle and sheep become infected by ingestion of grasses, plants, or water in damp or irrigated pastures where vegetation is infested with metacercariae. Man contracts the disease by ingesting plants such as watercress or lettuce containing the encysted metacercariae.

Ascaris ova are also spread from the feces of infected animals and man and are found in irrigation water (Wang and Dunlop 1954).[532] Cattle and hogs are commonly infected, where the adult worms mature in the intestinal tract, sometimes blocking the bile ducts. *Ascaris* ova have been reported to survive for 2 years in irrigated soil and have been found on irrigated vegetables even when chlorinated effluent was used for irrigation (Gaertner and Mueting 1951).[517]

Schistosomiasis, although not yet prevalent in the United States except in immigrants from areas where the disease exists, should be considered because infected individuals may move about the country and spread the disease. The life cycle of these schistosomes is similar to that of the liver fluke, in that eggs from the feces or urine of infected individuals are spread from domestic wastes and may reach surface irrigation water where the miracidial forms enter certain snails and multiply, releasing cercariae. Although these cercariae may produce disease if ingested by man, the more common method of infection is through the skin of individuals working in infested streams and irrigation ditches. Such infections are most common in Egypt (Barlow 1937)[511] and other irrigated areas where workers wade in the water without boots. It is unlikely that the cercariae would survive long on plants after harvest.

Little is known of the possibility that enteric viruses such as polioviruses, Coxsackie, ECHO, and infectious hepatitis viruses may be spread through irrigation practices. Murphy and his co-workers (Murphy et al. 1958)[527] tested the survival of polioviruses in the root environment of tomato and pea plants in modified hydroponic culture. In a second paper, Murphy and Syverton (1958)[528] studied the recovery and distribution of a variety of viruses in growing plants. The authors conclude that it is unlikely that plants or plant fruits serve as reservoirs and carriers of poliovirus. However, their findings of significant absorption of a mammalian virus in the roots of the plants suggest that more research is needed in this area.

Many microorganisms other than those specifically mentioned in this section may be transmitted to plants, animals, and humans through irrigation practices. One of the more serious of these is vibriosis. In some cases, definitive information on microorganisms is lacking. Although others, such as the cholera organisms, are significant in other parts of the world, they are no longer important in the United States.

Direct search for the presence of pathogenic microorganisms in streams, reservoirs, irrigation water, or on irrigated plants is too slow and cumbersome for routine control or assessment of quality. Instead, accepted index organisms such as the coliform group and fecal coli (Kabler et al. 1964),[521] which are usually far more numerous from these sources, and other biological or chemical tests, are used to assess water quality.

Recent studies have emphasized the value of the fecal coliform in assessing the occurrence of salmonella, the most common bacterial pathogen in irrigation water. Geldreich and Bordner (1971)[518] reviewed field studies involving irrigation water, field crops, and soils, and stated that when the fecal coliform density per 100 ml was above 1,000 organisms in various stream waters, *Salmonella* occurrence reached a frequency of 96.4 per cent. Below 1,000 fecal coliforms per 100 ml (range 1–1000) the occurence of *Salmonella* was 53.5 per cent.

Further support for the limit of 1,000 fecal coliforms per 100 ml of water is shown in the recent studies of Cheng et al. (1971),[514] who reported that as the fecal coliforms density reached less than 810 per 100 ml. downstream from a sewage treatment plant, *Salmonella* were not recovered.

Recommendation

Irrigation waters below the fecal coliform density of 1,000/100 ml should contain sufficiently low concentrations of pathogenic microorganisms that no hazards to animals or man result from their use or from consumption of raw crops irrigated with such waters.

THE USE OF WASTEWATER FOR IRRIGATION

An expanding population requires new sources of water for irrigation of crops and development of disposal systems for municipal and other wastewaters that will not result in the contamination of streams, lakes, and oceans. Irrigation of crops with wastewater will probably be widely practiced because it meets both needs simultaneously.

Wastewater From Municipal Treatment Systems

Various human and animal pathogens carried in municipal wastewater need to be nullified. Pathogens carried in municipal wastewater include various bacteria, spirochetes, helminths, protozoa, and viruses (Dunlop 1968).[538] Tanner (1944)[558] and Rudolfs et al. (1950)[555] have reviewed the literature on the occurrence and survival of pathogenic and nonpathogenic enteric bacteria in soil, water, sewage, and sludges, and on vegetation irrigated or fertilized with these materials. It would appear from these reviews that fruits and vegetables growing in infected soil can become contaminated with pathogenic bacteria and that these bacteria may survive for periods of a few days to several weeks or more in the soil, depending upon local conditions, weather, and the degree of contamination. However, Geldreich and Bordner (1971)[541] noted that pathogens are seldom detected on farm produce unless the plant samples are grossly contaminated with sewage or are observed to have fecal particles clinging to them. The level of pathogen recovery depends upon the incidence of waterborne disease in the area, the soil type, soil pH, soil moisture content, soil nutrient levels, antagonistic effects of other organisms, temperature, humidity, and length of exposure to sunlight.

Norman and Kabler (1953)[551] made coliform and other bacterial counts in samples of sewage-contamination river and ditch waters and of soil and vegetable samples in the fields to which these waters were applied. They found that although the bacterial contents of both river and ditch waters were very high, both soil and vegetable washings had much lower counts. For example, where irrigation water had coliform counts of 230,000/100 ml, leafy vegetables had counts of 39,000/100 grams and smooth vegetables, such as tomatoes and peppers, only 1,000/100 grams. High enterococcus counts accompanied high coliform counts in water samples, but enterococcus counts did not appear to be correlated in any way with coliform counts in soil and vegetable washings.

Dunlop and Wang (1961)[539] have also endeavored to study the problem under actual field conditions in Colorado. *Salmonella*, *Ascaris* ova, and *Entamoeba coli* cysts were recovered from more than 50 per cent of irrigation water samples contaminated with either raw sewage or primary-treated, chlorinated effluents. Only one of 97 samples of vegetables irrigated with this water yielded *Salmonella*, but *Ascaris* ova were recovered from two of 34 of the vegetable samples. Although cysts of the human pathogen, *Entamoeba histolytica*, were not recovered in this work, probably due to a low carrier rate in Colorado; their similar resistance to the environment would suggest that these organisms would also survive in irrigation water for a considerable period of time. It should be pointed out, however, that this work was done entirely with furrow irrigation on a sandy soil in a semiarid region, and the low recoveries from vegetables cannot necessarily be applied to other regions or to sprinkler irrigation of similar crops. In fact, Muller (1957)[550] has reported that two places near Hamburg, Germany, where sprinkler irrigation was used, *Salmonella* organisms were isolated 40 days after sprinkling on soil and on potatoes, 10 days on carrots, and 5 days on cabbage and gooseberries.

Muller (1955)[549] has also reported that 69 of 204 grass samples receiving raw sewage by sprinkling were positive for organisms of the typhoid-paratyphoid group (*Salmonella*). The bacteria began to die off 3 weeks after sewage application; but 6 weeks after application, 5 per cent of the samples were still infected. These findings emphasize the importance of having good quality water for sprinkler irrigation.

Tubercle bacilli have apparently not been looked for on irrigated crops in the United States. However, Sepp (1963)[557] stated that several investigations on tuberculosis infection of cattle pasturing on sewage-irrigated land have been carried out in Germany. The investigators are in general agreement that if sewage application is stopped 14 days before pasturing, there is no danger that the cattle will contract bovine tuberculosis through grazing. In contrast, Dedie (1955)[537] reported that these organisms can remain infective for 3 months in waste waters and up to 6 months in soil. The recent findings of a typical mycobacteria in intestinal lesions of cattle with concurrent tuberculin sensitivity in the United States may possibly be due to ingestion of these organisms either from soil or irrigated pastures.

Both animals and human beings are subject to helminth infections—ascariasis, fascioliasis, cysticerosis and tapeworm infection, and schistosomiasis—all of which may be transmitted through surface irrigation water and plants infected with the ova or intermediate forms of the organisms. The ova and parasitic worms are quite resistant to sewage treatment processes as well as to chlorination (Borts 1949)[533] and have been studied quite extensively in the application of sewage and irrigation water to various crops (Otter 1951,[553] Selitrennikova and Shakhurina 1953,[556] Wang and Dunlop 1954[560]). Epidemics have been traced to crop contamination with raw sewage but not to irrigation with treated effluents (Dunlop 1968).[538]

The chances of contamination of crops can be further reduced by using furrow or subirrigation instead of sprinklers, by stopping irrigation as long as possible before harvest begins, and by educating farm workers on sanitation practices for harvest (Geldreich and Bordner 1971).[541] It is better to restrict irrigation with sewage water to crops that are adequately processed before sale and to crops that are not used for human consumption.

Standards are needed to establish the point where irrigation waters that contain some sewage water must be restricted and to indicate the level to which wastewater must be treated before it can be used for unrestricted irrigation.

The direct isolation of pathogens is too slow and complicated for routine analyses of water quality (Geldreich and Bordner 1971).[541] A quantitative method for *Salmonella* detection has been developed recently (Cheng et al. 1971).[536] However, the minimum number of *Salmonella* required to cause infection are not known, and data are not available to correlate incidence of *Salmonella* with the incidence of other pathogens (Geldreich 1970).[540] The fecal coliform group has a high positive correlation with fecal contamination from warm-blooded animals and should be used as an indicator of pollution until more direct methods can be developed.

Information is available indicating the levels of fecal coliform at which pathogens can no longer be isolated from irrigation water. *Salmonella* were consistently recovered in the Red River of the north when fecal coliform levels were 1000/100 ml or higher, but were not detected at fecal coliform levels of 218 and 49/100 ml (ORSANCO Water Users Committee 1971).[552] Cheng et al. (1971)[536] reported numbers of fecal coliform at various distances downstream, and *Salmonella* was not isolated from samples containing less than 810 fecal coliforms/100 ml. Geldreich and Bordner (1971)[541] presented data from nationwide field investigations showing the relationship between *Salmonella* occurrence and fecal coliform densities. *Salmonella* occurrence was 53.5 per cent for streams with less than 1,000 fecal coliforms per 100 ml and 96.4 per cent for streams with more than 1,000 fecal coliforms per 100 ml. A maximum level of 1,000 fecal coliforms per 100 ml of water appears to be a realistic standard for water used for unrestricted irrigation.

Secondary sewage effluent can be chlorinated to reduce the fecal coliform bacteria below the 1,000 per ml limit, but viruses may survive chlorination. Wastewater used for unrestricted irrigation should receive at least primary and biological secondary treatment before chlorination. Filtration through soil is another effective way to remove fecal bacteria (Merrell et al. 1967,[548] Bouwer 1968,[534] Bouwer and Lance 1970,[535] Lance and Whisler 1972).[544]

The elimination of health hazards has been the primary consideration regulating the use of sewage water in the past. But control of nutrient loads must also be a prime concern. The nutrients applied to the land must be balanced against the nutrient removal capacity of the soil-plant system to minimize groundwater contamination. Kardos (1968)[542] reported that various crops removed only 20 to 60 per cent of the phosphorus applied in sewage water, but the total removal by the soil-plant system was about 99 per cent.

Many biological reactions account for nitrogen removal from wastewater, but heavy applications of sewage water can result in the movement of nitrogen below the root zone (Lance[543] *in press* 1972).

Work with a high-rate groundwater recharge system utilizing sewage water resulted in 30 per cent nitrogen removal from the sewage water (Lance and Whisler 1972).[544]

Nitrate can accumulate in plants supplied with nitrogen in excess of their needs to the point that they are a hazard to livestock. Nitrate usually accumulates in stems and leaves rather than in seeds (Viets 1965).[559]

The concentration of trace elements in sewage water used for irrigation should meet the general requirements established for other irrigation waters. Damage to plants by toxic elements has not yet been a problem on lands irrigated with sewage water in the United States. Problems could develop in some areas, however, if industries release potentially toxic elements such as zinc or copper into sewage treatment systems in large quantities. The concentration of boron in sewage water may become a problem if the use of this element in detergents continues to increase. The guidelines for salinity in irrigation water also apply to sewage water used for irrigation.

The organic matter content of secondary sewage water does not appear to be a problem limiting its use in irrigation. Secondary sewage effluent has been infiltrated into river sand at a rate of 100 meters per year in Arizona (Bouwer and Lance 1970).[535] The COD of this water was consistently reduced from 50 mg/l to 17 mg/l or the same COD as the native groundwater of the area. The organic load might be a factor in causing clogging of soils used for maximum irrigation to promote groundwater recharge. Suspended solids have not been reported to be a problem during irrigation with treated effluents.

Wastewater From Food Processing Plants and Animal Waste Disposal Systems

Wastewater from food processing plants, dairy plants, and lagoons used for treatment of wastes from feedlots, poultry houses, and swine operations, may also be used for irrigation. Some food processing wastewater is high in salt content and the guidelines for salinity control concerning unrestricted irrigation in the Section, Irrigation Quality for Arid Regions, should be followed (Pearson *in press* 1972[554]). Effluents from plants using a lye-peeling process are generally unsuitable for irrigation due to their high sodium content. All of the wastewaters mentioned above are usually much higher in organic content than secondary sewage effluent. This can result in clogging of the soil surface, if application rates are excessive (Lawton et al. 1960,[547] Law 1968,[545] Law et al. 1970).[546] Only well drained soils should be irrigated, and runoff should be prevented unless a closely managed spray-runoff treatment system is used. The nutrient content of the wastewaters varies considerably. The nutrient load applied should be balanced against the nutrient removal capacity of the soil. Food processing wastes present no pathogenic problem and may be used for unrestricted irrigation. Since some animal pathogens also infect humans, water containing animal wastes should not be applied with sprinkler systems to crops that are consumed raw.

Recommendations

- **Raw sewage should not be used in the United States for irrigation or land disposal.**
- **Sewage water that has received primary treatment may be used on crops not used for human consumption. Primary effluents should be free of phytotoxic materials.**
- **Sewage water that has received secondary treatment may also be used to irrigate crops that are canned or similarly processed before sale.**
- **Fecal coliform standard for unrestricted irrigation water should be a maximum of 1,000/100 ml.**
- **The amount of wastewater that can be applied is determined by balancing the nutrient load of the wastewater against the nutrient removal capacity of the soil.**
- **Phosphorus will probably not limit sewage application because of the tremendous adsorption capacity of the soil.**
- **The nitrogen load should be balanced against crop removal within 30 per cent unless additional removal can be demonstrated.**

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INTRODUCTION

WATER USE

Since the advent of the industrial era, the use and availability of water has been of primary concern to industry both in the selection and design of plant sites and in plant operation. By 1968 the water withdrawal of industry—including both industrial manufacturing plants and investor-owned thermal electric utilities—had reached a total of approximately 84,000 billion gallons per year (bgy). Of these, about 93 per cent or 78,000 bgy was used for cooling or condensing purposes; 5 per cent or nearly 4,300 bgy was used for processing, including water that came in contact with the product as steam or coolant; and less than 2 per cent or 1,000 bgy was used as boiler-feed water (U. S. Department of Commerce, Bureau of the Census 1971[19] hereafter referred to as Bureau of the Census 1971).[5] *

Of the total intake nearly 30 per cent or 25,000 bgy was brackish water containing more than 1,000 milligrams per liter (mg/l) of dissolved solids. The freshwater intake amounted to 59,000 bgy; 56,000 of these took the form of surface water delivered by water systems owned by the user company. Groundwater amounted to 2,300 bgy, a relatively small percentage of the total intake, but its significance and importance cannot be overlooked in view of the number of industrial plants that use it for part or all of their supply.

Thirty per cent of the approximately 4,000 bgy used by the manufacturing processes in 1968 was treated or secured from a public water supply. Ninety per cent of all the water the manufacturing industry used for boiler feed and processing was represented in this figure. Water for cooling or condensing represented over 90 per cent of total industrial water use. The largest part of this was on a once-through basis where only a minimum of treatment was economically feasible.

Table VI-1 summarizes the information on water intake, recycling, and consumption for each industrial group considered in this Section. Recycling may include reuse for different cooling or process systems, recirculation through cooling towers or cooling ponds, recharge of water to an underground aquifer, or reuse of effluents from sewage or waste treatment plants.

TABLE VI-1—Industrial Plant and Investor-Owned Thermal Electric Plant Water Intake, Reuse, and Consumption, 1968

SIC	Industrial group	Water intake (bgy)				Water recycled (bgy)	Gross water use, including recycling (bgy)	Water consumed (bgy)	Water discharged (bgy)
		Purpose							
		Cooling and condensing	Boiler feed, sanitary service, etc.	Process	Total				
20	Food and kindred products	427	93	290	810	535	1,345	57	753
22	Textile mill products	24	22	109	155	174	329	19	136
24	Lumber and wood products	62	20	37	119	87	206	26	93
26	Paper and allied products	652	123	1,478	2,253	4,270	6,523	175	2,078
28	Chemicals and allied products	3,533	210	733	4,476	4,940	9,416	301	4,175
29	Petroleum and coal products	1,230	111	95	1,436	5,855	7,291	219	1,217
31	Leather and leather products	1	1	14	16	4	20	1	15
33	Primary metal industry	3,632	165	1,207	5,004	2,780	7,784	308	4,696
	Subtotal	9,561	745	3,963	14,269	18,645	32,914	1,106	13,163
	Other Industries	574	291	332	1,197	1,589	2,786	84	1,113
	Total Industry	10,135	1,036	4,295	15,466	20,234	35,700	1,190	14,276
	Thermal electric plants	68,200	(a)		68,200	8,525	76,725	100	68,100
	TOTAL	78,335	1,036 (b)	4,295	83,666	28,759	112,425	1,290	82,376

[a] Boiler-feed water use by thermal electric plants estimated to be equivalent to industrial sanitary service, etc., water use.

[b] Total boiler-feed water.

Bureau of the Census 1971[5]

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* Literature citations appear at the end of the Section. They can be located alphabetically or by superscript number.

SCOPE

After describing industry's use of water in steam generation and cooling, the panel on Industrial Water Supplies examined ten groups of one or more industries as defined by the Standard Industrial Classification (SIC) coding used by the Bureau of the Census (U. S. Executive Office of the President, Bureau of the Budget 1967).[22]

The industries included textile mills (SIC 22), lumber and wood (SIC 24), pulp and paper (SIC 26), chemical and allied products (SIC 28), petroleum refining (SIC 2911), primary metals (SIC 33), food canning (SIC 2032 and 2033), bottled and canned soft drinks (SIC 2086), tanning (SIC 3111), and mining and cement (SIC 10). Only the major users of water were included, representing a variety of industries in order to insure that a wide cross section of water qualities would be described.

Industrial effluents cause water quality changes in the receiving systems, but consideration of these changes was not part of the charge to the Panel on Industrial Water Supplies. The other Sections in this Report include consideration of the effects of many specific constituents of such effluents as related to various water uses.

WATER QUALITY REQUIREMENTS

Water quality requirements differ widely for the broad variety of industrial uses, but modern water treatment tech-

TABLE VI-2—Summary of Specific Quality Characteristics of Surface Waters That Have Been Used as Sources for Industrial Water Supplies

(Unless otherwise indicated, units are mg/l and values are maximums. No one water will have all the maximum values shown)

Characteristics	Boiler Makeup water		Cooling Water				Process Water									
			Fresh		Brackish[a]								Mining Industry		Oil Recovery Injection Waters	
	Industrial 0 to 1,500 psig	Utility 700 to 5,000 psig	Once through	Makeup recycle	Once through	Makeup recycle	Textile Industry SIC-22	Lumber Industry SIC-24	Pulp and Paper Industry SIC-26	Chemical Industry SIC-28	Petroleum Industry SIC-29	Prim. Metals Industry SIC-33	Copper Sulfide Concentrator for Process Water	Copper Leach Solution	Sea Water	Formation Water
Silica (SiO_2)	150	150	50	150	25	25			50		85					
Aluminum (Al)	3	3	3	3										12,000		
Iron (Fe)	80	80	14	80	1.0	1.0	0.3		2.6	10	15			12,000[g]	0.2	13
Manganese (Mn)	10	10	2.5	10	0.02	0.02	1.0			2						
Copper (Cu)							0.5									
Calcium (Ca)			500	500	1,200	1,200				250	220		1,510 ($CaCO_3$)		400	2,727
Magnesium (Mg)										100	85			12,000	1,272	655
Sodium & potassium (Na+K)											230				10,840	42,000
Ammonia (NH_3)											40					
Bicarbonate (HCO_3)	600	600	600	600	180	180				600	480				142	281
Sulfate (SO_4)	1,400	1,400	680	680	2,700	2,700				850	900		1,604	64,000	2,560	42
Chloride (Cl)	19,000	19,000	600	500	22,000	22,000			200[b]	500	1,600	500	12		18,980	72,782
Fluoride (F)											1.2					
Nitrate (NO_3)			30	30							8					
Phosphate (PO_4)		50	4	4	5	5										
Dissolved Solids	35,000	35,000	1,000	1,000	35,000	35,000	150		1,080	2,500	3,500	1,500	2,100		34,292	118,524
Suspended Solids	15,000	15,000	5,000	15,000	250	250	1,000	(c)		10,000	5,000	3,000				
Hardness ($CaCO_3$)	5,000	5,000	850	850	7,000	7,000	120		475	1,000	900	1,000	1,530			
Alkalinity ($CaCO_3$)	500	500	500	500	150	150				500	500	200	415			
Acidity ($CaCO_3$)	1,000	1,000	0	200	0	0						75				
pH, units			5.0-8.9	3.5-9.1	5.0-8.4	5.0-8.4	6.0-8.0	5-9	4.6-9.4	5.5-9.0	6.0-9.0	3-9	to 11.7	3-3.5	...	to 6.5
Color, units	1,200	1,200		1,200					360	500	25					
Organics:																
Methylene blue active substances	2[d]	10	1.3	1.3		1.3										
Carbon tetrachloride extract	100	100	(e)	100	(e)	100						30				
Chemical oxygen demand (COD)	100	500		100		200										
Hydrogen sulfide (H_2S)					4	4					20					
Temperature, F	120	120	100	120	100	120			95[f]			100				

[a] Water containing in excess of 1,000 mg/l dissolved solids.
[b] May be ≤ 1,000 for mechanical pulping operations.
[c] No particles ≥ 3 mm diameter.
[d] One mg/l for pressures above 700 psig.
[e] No floating oil.
[f] Applies to bleached chemical pulp and paper only.
[g] 12,000 mg/l Fe includes 6,000 Fe^{+}; and 6,000 Fe^{++}.
ASTM Standards 1970 or Standard Methods 1971[16]

nology is capable of treating almost any raw water to render it suitable for any industrial use. The treatment may be costly, and may require large ground space not always available at otherwise suitable plant locations. Sometimes the substitution of a more expensive alternative supply is necessary. Nevertheless, in most cases, the costs involved are but a small part of the total production and marketing costs of the industrial product in question.

It is evident that the more nearly the composition of an available water supply approaches the particular composition needed, the more desirable that water is, and, conversely, the more such compositions differ, the more difficult and expensive it is to modify the water for use. Improper operation or malfunction of control instruments or water treating equipment may cause a deterioration of the treated water, and this, in turn, can cause deterioration or loss of product and damage to equipment. The poorer the quality of the raw water, the more serious the consequences of such malfunctions.

Improving the quality of a given water supply will only incrementally decrease the cost of treatment for an industrial installation, because it is often too late to make economical alterations in the existing water treatment facilities. For the same reason, if the quality characteristics of the water supply are allowed to deteriorate from their usual range, the cost for treatment can be substantially increased. On the other hand, improved water supply characteristics at a given site may mean lower water treatment costs for other industries subsequently established there.

Table VI-2 summarizes quality characteristics of surface waters at the point of intake that have been used as sources of boiler makeup, cooling, and process water.

CONCLUSIONS

- Industry is diversified in kind, size, and product. It incorporates many processes, including different ones to achieve the same ends. Water quality requirements for different industries, for various industrial processes within a single plant, and for the same process in different plants vary widely.
- Water quality requirements at *point of use*, as distinguished from requirements at *point of intake*, are established for a number of industrial processes but are inadequately defined or nonexistent for others.
- Modern water treatment technology permits water of virtually any quality to be treated to provide the characteristics desired by industry at point of use. Occasionally, this may be costly; but in general the cost of treating water for specific processes is acceptable to industry, because it is only a small part of total production and marketing costs.
- Although water quality at *point of use* is critical for many industrial processes, industry's *intake water* quality requirements are not as stringent as those for public water supplies, recreational or agricultural use, or support of aquatic life.
- Because of the diversity of industrial water quality requirements, it is not possible to state specific values for intake water quality characteristics for industrial use. Ordinarily these values lie between those that have been used by industries for sources of water (Table VI-2) and the quality recommended for other uses in other sections of this book.

Recommendations

Desirable intake water quality characteristics for industrial water supplies can be meaningfully designated as a range lying between the values that have been used by industry for sources of water (Table VI–2) and the quality characteristics recommended for other water uses in other chapters of this Section. Values that exceed those in Table VI–2 would ordinarily not be acceptable to industry.

BASIC WATER TREATMENT PROCESSES

A wide range of treatment processes is available to produce water of the required quality for industry at the point of use. Treatment methods fall into two general categories: external and internal. External treatment refers to processes utilized in altering water quality prior to the point of use. The typical household water softening unit is an external treatment. Internal treatment refers to processes limited basically to chemical additives utilized to alter water quality at the point of use or within the process. Water softening compounds used in laundering are forms of internal treatment. Water treatment processes are in themselves users of water. Normally, 2 to 10 per cent of the feed water ends up as waste generated by treatment processes (see Table VI-3). Thus, the actual water intake is greater than the treated water produced.

EXTERNAL WATER TREATMENT PROCESSES

Figure VI-1 is a schematic diagram of the most common external water treatment processes. Properly applied, alone or in various combinations, these processes can convert any incoming water quality to a usable quality. A dramatic example is the conversion of brackish water to a water that exceeds the quality of distilled water.

Note that the flow chart illustrates many processes and that a particular process is applied to remove a particular contaminant. If that contaminant does not appear in the water or is harmless for the intended use of the water, that process would not be used. For example, a clear well water might not need filtration prior to further treatment. In addition, the water use determines the extent of treatment. For example, to use Mississippi River water for cooling, rough screening to remove the floating debris may be sufficient for some applications, whereas clarification and filtration may be required for other uses. To use that same water for makeup for a super critical pressure boiler would require further treatment by ion exchange, perhaps strong cation, strong anion, and mixed bed exchangers.

As previously stated, industry's need for water can be met even under the poorest conditions. However, the use of water treatment systems is not without consequence. External water treatment processes concentrate a particular contaminant or contaminants. Thus, in the quest for pure water, a waste product is generated. The waste product is a pollutant and the cost of its disposal must be considered as part of the overall cost of water treatment.

The estimates of waste volume and solids in Table VI-3 are based on treating a water with an analysis such as shown in Table VI-4. Table VI-4 also illustrates an analysis of several common forms of water treatment. The estimates are thus typical only of the water described and will vary with different water supplies. Waste volumes are stated as a percentage of inlet flow. Thus, a 2,000 gallon per minute (gpm) clarifier will discharge 40 to 100 gpm of sludge.

The following paragraphs briefly describe the available treatment methods, outline their capabilities, and combined with Table VI-3, provide a general idea of the waste produced. (The groupings A, B, and C do not imply treatment schemes or necessarily indicate a sequence of treatment.) The processes are applicable to various water characteristics; it is immaterial whether the supply is surface or ground water. Since the equipment used can be of appreciable size, available land area can be an important factor in the selection of a particular process.

Group A Processes

Rough Screens Generally installed at the actual point of intake, rough screens are simple bars or mesh screens used to trap large objects and prevent damage to pumps and other mechanical equipment.

Sedimentation This process takes place in large open basins used to reduce the water velocity so that heavier suspended particles can settle out.

Clarification Chemical additives (e.g., aluminum salts, iron salts, lime) are used in large open basins so that practically all suspended matter, color, odors, and organic compounds can be removed efficiently.

Lime Softening (cold) The equipment used here is similar to that used for clarification. In addition to flocculent chemicals, lime and sometimes soda ash are used in large open basins. Clarification is obtained, and a large portion of the calcium and magnesium bicarbonates are removed.

Lime Softening (hot) The process is, in general, the same as cold except that it is carried out at or above 212 F. The results are the same but with the added benefit of silica removal. The characteristics of wastes are the same but at a high temperature. Note that further treatment of hot lime effluent is generally limited to filtration and sodium cation exchange.

TABLE VI-3—Waste Generated by Treatment Processes

Treatment process[a]	Character of waste produced	Waste volume percentage flow	Example of waste weight[b] dry basis pounds solids 1,000 gal processed
Rough screens	Large objects, debris		
Sedimentation	Sand, mud slurry	5-10	
Clarification	Usually acidic chemical sludge and settled matter	2-5	1.3
Cold lime softening	Alkaline chemical sludge and settled matter	2-5	1.7
Hot lime softening (+212 F)	Alkaline chemical sludge and settled matter	2-5	1.7
Aeration	Gaseous, possible air pollutant, such as hydrogen sulfide		
Filtration, gravity, or pressure	Sludge, suspended solids	2-5 (for packed bed units)	0.1-0.2
Adsorption, activated carbon for odors, tastes, color, organics	Exhausted carbon if not regenerated. Small amounts carbon fines and other solids can appear in backwash. Carbon regeneration is separate process (usually thermal) in which air pollution problems must be met.	2-5	
Manganese zeolite, for iron removal	Iron oxide suspended solids	Similar to other filtration prodesses	
Miscellaneous, e.g., precoat, membrane, dual media filtration fine straining	As in other filters. Precoat waste includes precoat materials.	1-5	0.1-0.2 (plus precoat materials when used)
Reverse osmosis[c]	Suspended and 90-99 percent of dissolved solids plus chemical pretreatment if required	10-50	1.0-2.0
Electrodialysis[c]	Suspended and 80-95 percent of dissolved solids plus chemical pretreatment if required	10-50	1.0-2.0
Distillation	Concentrated dissolved and suspended solids	10-75	1.5
Ion exchange processes[d]			
Sodium cation	Dissolved calcium, magnesium and sodium chlorides	4-6	1.3
2-bed demineralization	Dissolved solids from feed plus regenerants	10-14	4-5
Mixed bed demineralization	Dissolved solids from feed plus regenerants	10-14	>5
Internal processes	Chemicals are added directly into operating cycle. At least a portion of process steam containing added chemicals, dissolved and suspended solids from feed, and possibly contamination from process can be extracted from the cycle for disposal or treatment and recycle.		

[a] Processes are used alone or in various combinations, depending upon need.

[b] Amounts based on application of process to raw water shown in Table VI-4. These values do not necessarily apply when these processes are used in combinations.

[c] Feed must be relatively free of suspended matter.

[d] There are many variations. Listed here are a few of the most important.

TABLE VI-4—Typical Raw Water Analyses and Operating Results (mg/l, unless otherwise indicated)

Constituent	Expressed as	Raw water[a]	After clarification and filtration	After cold lime softening and filtration	After clarification, filtration, and sodium-cation exchange softening	After clarification, filtration, and demineralization
Cations[a]						
Calcium	$CaCO_3$	51.5	51.5	38.7	1.0	0
Magnesium	"	19.5	19.5	17.5	1.0	0
Sodium	"	18.6	18.6	18.6	87.6	1-2
Potassium	"	1.8	1.8	1.8	1.8	0
Total Cations	"	91.4	91.4	76.6	91.4	1-2
Anions[a]						
Bicarbonate	"	56.8	47.8	0	47.8	0
Carbonate	"	0	0	33.0	0	0
Hydroxide	"	0	0	0	0	1-2
Sulfate	"	21.8	30.8	30.8	30.8	0
Chloride	"	12.0	12.0	12.0	12.0	0
Nitrate	"	0.8	0.8	0.8	0.8	0
Total Anions	"	91.4	91.4	76.6	91.4	1-2
Iron[a]	Fe	0.16	Nil	Nil	Nil	Nil
Silica[a]	SiO_2	9.0	9.0	9.0	9.0	0.01
Color[b]	units	15.0	2-5	2-5	Nil	Nil
Turbidity[b]	"	100.0	0-2	0-2	Nil	Nil
pH[a]	"	6.5-7.5	6.0-8.0	9.0-11.0	6.0-8.0	7.0-9.0

[a] Taken from Livingstone 1963[a]; adjusted slightly for ion balance and for expression as $CaCO_3$ equivalents.

[b] Developed by the Panel for illustrative purposes.

Aeration This process, which can be in several different physical forms, is applied to reduce the concentration of carbon dioxide, thereby reducing the chemicals required for cold lime softening. Aeration oxidizes iron and manganese to allow their removal by clarification, lime softening, or filtration. No solid wastes flow from an aerator, but released gases such as hydrogen sulfide can present a problem.

Miscellaneous There are other special variations of all the primary treatment methods that can be applied under specific circumstances.

Group B Processes

Filtration This process uses gravity or pressure units in which traces of suspended matter are removed by passage through a bed of sand, anthracite coal, or other granular material. In general, the effluent at the primary stage must be filtered prior to further treatment. Some waters can be filtered without primary treatment. A filter is cleaned by reversing the direction of the water flow (backwashing).

Adsorption This is a separation process designed primarily to remove dissolved organic materials including odor, taste, and color-producing compounds. Activated carbon is generally used for this purpose. Backwashing of fixed adsorption units produces small amounts of solids as the feed has generally been filtered prior to passage over the carbon. Expanded bed adsorption units do not require regular backwashing. Chemical or thermal regeneration of

(Items not enclosed in boxes indicate typical water uses for treatment methods shown.)

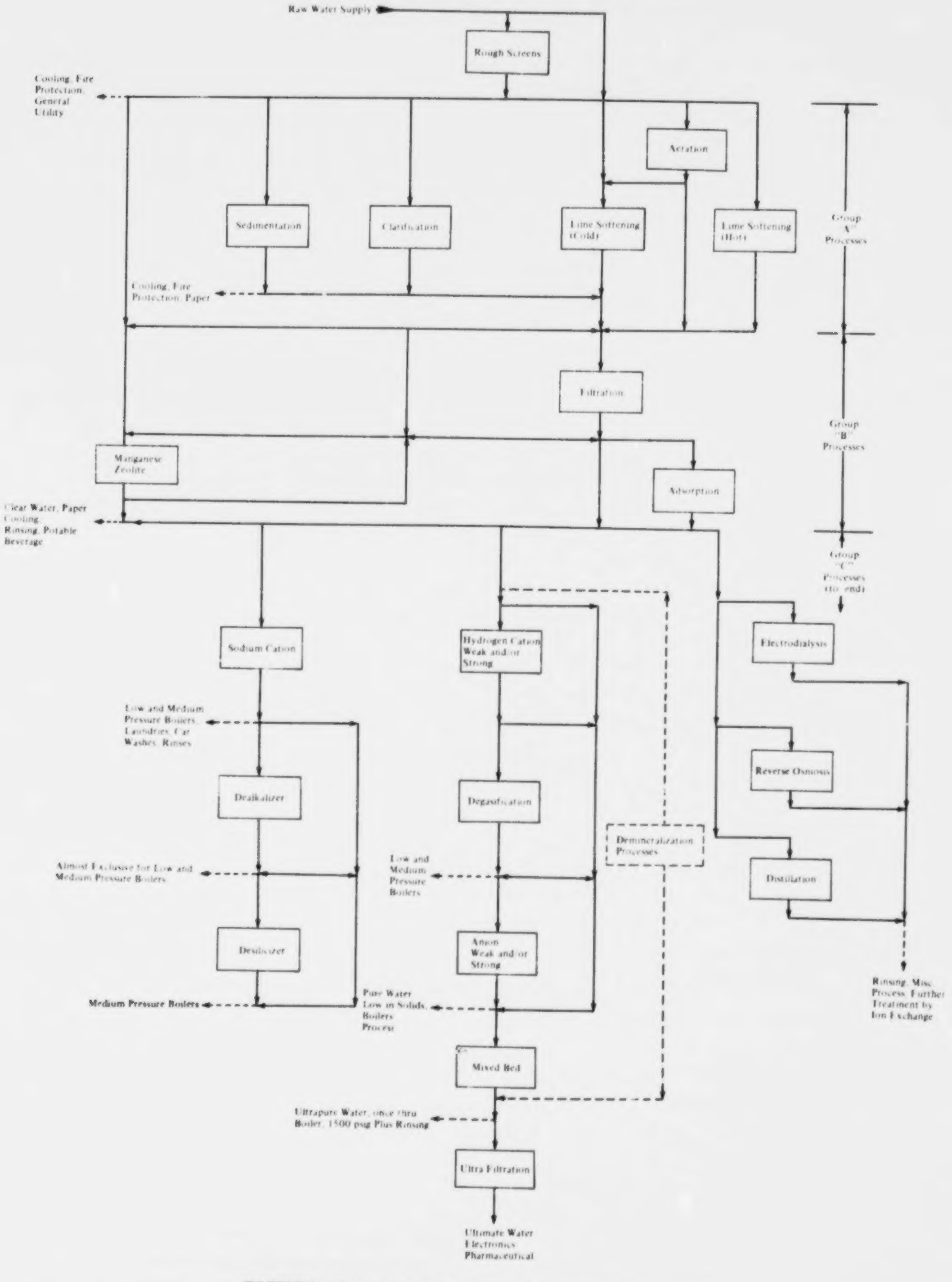
Raw Water Supply
Rough Screens
Cooling, Fire Protection, General Utility
Aeration
Sedimentation
Clarification
Lime Softening (Cold)
Lime Softening (Hot)
Group "A" Processes
Cooling, Fire Protection, Paper
Filtration
Group "B" Processes
Manganese Zeolite
Adsorption
Clear Water, Paper Cooling, Rinsing, Potable Beverage
Group "C" Processes (to end)
Sodium Cation
Hydrogen Cation Weak and/or Strong
Electrodialysis
Low and Medium Pressure Boilers, Laundries, Car Washes, Rinses
Dealkalizer
Degasification
Reverse Osmosis
Demineralization Processes
Almost Exclusive for Low and Medium Pressure Boilers
Low and Medium Pressure Boilers
Distillation
Desilicizer
Anion Weak and/or Strong
Medium Pressure Boilers
Pure Water Low in Solids, Boilers Process
Rinsing, Misc Process, Further Treatment by Ion Exchange
Mixed Bed
Ultrapure Water, once thru Boiler, 1500 psig Plus Rinsing
Ultra Filtration
Ultimate Water Electronics Pharmaceutical

FIGURE VI-1—External Water Treatment Processes

(Items not enclosed in boxes indicate typical water uses for treatment methods shown.)

carbon can remove adsorbed impurities and restore adsorptive efficiency and capacity.

Manganese Zeolite This process, specifically used for iron removal, is a special combined form of oxidation and filtration with a feed of potassium permanganate.

Miscellaneous Many specialized forms applicable to specific conditions are available. These include precoated filters, membrane filters, strainers, and dual media filters.

Group C Processes

Ultrafiltration Various types of pressure filters including membranes, cartridges, and discs can remove suspended solids larger than 0.1 to 1.0 micron, depending on the application.

Reverse Osmosis This relatively new development uses high pressures to force water through a membrane, preventing the passage of all suspended matter and up to 90–99 per cent of dissolved solids. The product water can be used directly or may require further treatment by ion exchange. The influent must be essentially free of suspended solids.

Electrodialysis A relatively new development, this process uses cationic and anionic membranes with applied direct current to remove dissolved solids. The product water can be used directly or may require further treatment by ion exchange. The feed must be essentially free of suspended matter.

Distillation This process uses thermal evaporation and condensation of water so that the condensate is free of suspended solids and 98–99 per cent of the dissolved solids are removed. Certain conditions may require the addition of special chemicals. The product water can be used directly or may require further treatment by ion exchange. The feed must be relatively free of suspended matter.

Ion Exchange Ion exchange is a versatile process with several dozen variations. Ion exchange technology is rapidly advancing. New resins, regeneration techniques, and operation modes are being introduced. Some of the more common applications are shown in Table VI-3. The exact arrangement of an ion exchange system depends upon raw water quality, desired treated water quality, flow rate, and economics. Total demineralization can remove in excess of 99 per cent of dissolved solids with feeds as high as 2,000 parts per million (ppm) or more. The waste produced by an ion exchanger includes the backwash and rinse waters, the regeneration effluent containing the exchanged ions, and the excess regenerative chemical. In general, the feed to any ion exchanger should contain no or only small quantities of suspended matter, color, and organics.

Cation Cation exchange removes cations from the water and replaces them with other cations from an ion exchanger. When in the hydrogen or acid form, strong cation (i.e., strong acid) can exchange hydrogen ions for the cations of either weak or strong acids, whereas weak cation (i.e., weak acid) exchanges hydrogen only for that fraction of cations equivalent to the weakly acidic anions present, such as bicarbonate.

Sodium Cation This is the simplest form of ion exchange. Sodium ions are exchanged for hardness ions (e.g., calcium, magnesium).

Anion Anion exchange removes anions from the water and replaces them with other anions from the ion exchanger. When in the base form, strong anion exchangers are capable of exchanging hydroxyl ions for the anions of either weak or strong acids, whereas weak anion exchangers exchange only with anions of strong acids.

Demineralization In industrial water treatment, demineralization refers to a sequence of cation exchange in which hydrogen ions are substituted for other cations followed by anion exchange in which hydroxyl ions are substituted for other anions. The product is H^+ plus OH^-; i.e., water.

Mixed Bed Mixed bed exchange provides complete demineralization in one step by the use of an intimate mixture of cation and anion resin in one unit. It is generally used for the polishing service step of high purity water. A cation-anion exchange system might produce a water containing 1.0 ppm of dissolved solids. After treatment by mixed bed, the solids would be down as low as 0.01 ppm.

Miscellaneous There are several specialty ion exchangers including: dealkalizers—chloride anion exchange for the removal of alkalinity; desilicizers—hydroxide anion exchange for the removal of silica (without previous hydrogen cation). Degasification equipment is used to remove carbon dioxide in order to reduce the work of the strong anion units that follow.

INTERNAL WATER TREATMENT PROCESSES

Internal water treatment processes are numerous. They include the addition of acid and alkali for pH control; polyphosphates, phosphonates, or polyelectrolytes for scale control; polymers for dispersal of sediment; phosphates and alkali for precipitation of hardness; amines, chromates, zinc, or silicates for corrosion control; sulfites or hydrazine for oxygen scavenging; and polyphosphates for sequestration of iron or manganese. Here again, the chemical feed is determined by the requirements. The industrial user produces the water quality that is needed, but a problem can be created when the user must dispose of all or part of the treated water. The choice of chemicals added to water must be considered in light of their potential as pollutants.

MAJOR INDUSTRIAL USES OF WATER

STEAM GENERATION AND COOLING

Description of the Industry

Steam generation and cooling are required in most industries. Waters used for these purposes are in Standard Industry Classifications 20 through 39 (with the exception of 23 and 27), plus the electric utility industry and mining (U. S. Executive Office of the President, Bureau of the Budget 1967).[22] (Water used as makeup for generation of steam that comes into direct contact with a product and cooling water that comes into direct contact with a product were considered to be process waters and, therefore, were not included in this Section.)

Both steam generation and cooling are encountered under a wide variety of conditions that require a correspondingly broad range of water quality recommendations. For example, steam may be generated in boilers that operate at pressures ranging from less than 10 pounds per square inch gauge (psig) for space heating to more than 3,500 psig for electric power generation. For any particular operating pressure, the required boiler water quality recommendations depend upon many factors in addition to the water temperature in the steam generator. Thus, the amount of potentially scale-forming hardness present in the makeup water to a low pressure boiler is of far less importance when the steam is used for space heating than when it is used for humidification of air. In the first case, virtually all of the steam is returned to the boiler as condensate so that there is only limited change in the amount of potential scale. In the second case, no condensate returns to the boiler so that scale-forming salts entering with the makeup water are concentrated.

The general recommendations for water to be used for boiler feed water could not be applied directly to an individual boiler without consideration of boiler design, operating practices, operating temperatures and pressures, makeup rates, and steam uses. All of these affect the nature of water-caused problems that might be anticipated in a boiler and its auxiliaries. These statements apply equally to water at source and at point of use.

Most high pressure boiler plants (Table VI-5) use some form of ion exchange in treatment of water for boiler feed. A few components of raw waters can cause abnormal difficulties and expense in these treatment plants Large organic molecules may block the exchange groups of the ion exchange resins and cannot be removed by normal regeneration procedures. Oily matter, especially of petroleum origin, will irreversibly coat ion exchange materials and filter media. Certain forms of silica may also block ion exchange resins irreversibly. Strong oxidants in polluted water have been known to destroy ion exchange resins in a surprisingly short time. Although most of these problems can be solved by available pretreatment methods, the equipment needed for such treatment may require more space than is available. This is especially true in industrial plants located in cities.

Cooling water uses are similarly diverse. They may be once-through or recirculated. Once-through cooling waters are drawn from amply large sources such as rivers, lakes, estuaries, or the sea. They are returned to these sources or to other large bodies of water after having passed through heat exchange equipment just once. The quantities of water required for once-through cooling are so huge that it is rarely economically feasible to alter their quality by treatment. Therefore, when a plant uses water for cooling on a once-through basis, the construction materials for the cooling system must be selected to withstand corrosion by the water available at the site. In such cases, the quality, as well as quantity, of available water may affect plant site selection.

The treatments commonly applied to once-through cooling waters are (a) screening for removal of debris, plants, or fish that can interfere with water flow, and (b) chlorination for control of biological organisms that interfere with water flow or heat transfer and contribute to localized corrosion. A few components of the intake water have been known to cause catastrophic failures in once-through cooling equipment. Damaging substances include hydrogen sulfide, oil, and suspended solids. Particularly pernicious are plastic containers usually originating from garbage disposal operations, or sheets of flexible plastic that can pass through a pump and then spread across a tube sheet in-

TABLE VI-5—*Quality Requirements of Water at Point of Use for Steam Generation and Cooling in Heat Exchangers*

(Unless otherwise indicated, units are mg/l and values that normally should not be exceeded. No one water will have all the maximum values shown.)

Characteristic	Boiler feedwater				Cooling water			
	Quality of water prior to the addition of chemicals used for internal conditioning							
	Industrial			Electric utilities	Once through		Makeup for recirculation	
	Low pressure 0 to 150 psig	Intermediate pressure 150 to 700 psig	High pressure 700 to 1,500 psig	1,500 to 5,000 psig	Fresh	Brackish[a]	Fresh	Brackish[a]
Silica (SiO_2)	30	10	0.7	0.01	50	25	50	25
Aluminum (Al)	5	0.1	0.01	0.01	(*b*)	(*b*)	0.1	0.1
Iron (Fe)	1	0.3	0.05	0.01	(*b*)	(*b*)	0.5	0.5
Manganese (Mn)	0.3	0.1	0.01	0.01	(*b*)	(*b*)	0.5	0.02
Calcium (Ca)	(*b*)	0.4	0.01	0.01	200	420	50	420
Magnesium (Mg)	(*b*)	0.25	0.01	0.01	(*b*)	(*b*)	(*b*)	(*b*)
Ammonia (NH_4)	0.1	0.1	0.1	.07	(*b*)	(*b*)	(*b*)	(*b*)
Bicarbonate (HCO_3)	170	120	48	0.5	600	140	24	140
Sulfate (SO_4)	(*b*)	(*b*)	(*b*)	(*d*)	680	2,700	200	2,700
Chloride (Cl)	(*b*)	(*b*)	(*b*)	(*b, d*)	600	19,000	500	19,000
Dissolved solids	700	500	200	0.5	1,000	35,000	500	35,000
Copper (Cu)	0.5	0.05	0.05	0.01	(*b*)	(*b*)	(*b*)	(*b*)
Zinc (Zn)	(*b*)	0.01	0.01	0.01	(*b*)	(*b*)	(*b*)	(*b*)
Hardness ($CaCO_3$)	350	1.0	0.07	0.07	850	6,250	650	6,250
Alkalinity ($CaCO_3$)	350	100	40	1	500	115	350	115
pH, units	7.0–10.0	8.2–10.0	8.2–9.0	8.8–9.4	5.0–8.3	6.0–8.3	(*b*)	(*b*)
Organics:								
Methylene blue active substances	1	1	0.5	0.1	(*b*)	(*b*)	1	1
Carbon tetrachloride extract	1	1	0.5	(*b, c*)	(*e*)	(*e*)	1	2
Chemical oxygen demand (COD)	5	5	1.0	1.0	75	75	75	75
Hydrogen sulfide (H_2S)	(*b*)	(*b*)	(*b*)	(*b*)	......	......	(*b*)	(*b*)
Dissolved oxygen (O_2)	2.5	0.007	0.007	0.007	present	present	(*b*)	(*b*)
Temperature, F	(*b*)	(*b*)	(*b*)	(*b*)	(*b*)	(*b*)	(*b*)	(*b*)
Suspended solids	10	5	0.5	0.05	5,000	2,500	100	100

[a] Brackish water—dissolved solids more than 1,000 mg/l by definition 1963 Census of Manufacturers.

[b] Accepted as received (if meeting other limiting values); has never been a problem at concentrations encountered.

[c] Zero, not detectable by test.

[d] Controlled by treatment for other constituents.

[e] No floating oil.

ASTM 1970 or Standard Methods 1971[16]

stantaneously shutting off a substantial part of the cooling water flow.

Treatment of once-through cooling waters drawn from underground aquifers is further limited if the water is conserved by return to an aquifer through recharge wells. In such cases treatment must not create changes that can cause clogging of the return aquifer.

When cooling ponds are used for heat rejection, the economics of water treatment are similar to those encountered with once-through cooling waters. On the other hand, most recirculating cooling water systems utilize cooling towers, and in these the water withdrawn from surface, ground, or municipal sources is small in comparison with the rate of circulation through the heat transfer equipment. Under these conditions, water treatment is economically feasible. Indeed, it becomes a necessity because of the changes in water composition produced by evaporation, air scrubbing, and other processes occuring during recirculation.

As in the case of steam generation, there is such a great variety of materials and operating conditions encountered in industrial heat exchange equipment, such a wide range of chemical and physical changes that can take place in the recirculated cooling water, and such a variety of water treatment and conditioning methods, that quality recommendations for makeup water for recirculating cooling systems can have only very limited practical significance. The needs of any specific system must be established on the basis of the makeup water composition and the construction and operating characteristics of each system. In general, the lower the hardness and alkalinity of the water supply, the more acceptable it is for cooling tower makeup.

Processes Utilizing Water

Steam Generation In 1968, manufacturing plants used about 1,036 billion gallons of water for boiler feed (makeup), sanitary service, and uses other than process or cooling (Bureau of the Census 1971).[5] No basis is given for a breakdown of this figure into its components, but boiler feed is the largest part.

Boiler makeup requirements of steam electric powerplants are small compared with their cooling water requirements. They are estimated to be only about 0.3 million gallons per day for a 1 million kilowatt plant operating at full load (Water Resources Council 1968).[24]

Based on the 1970 figures of 281 million kilowatts capacity of steam electric plants, a maximum of about 31 billion gallons of water was the total intake for steam generation in these plants (Edison Electric Institute *personal communication* 1970).[25] It is estimated that this quantity approximates the "sanitary service and other uses" in the industrial requirements, so that of the 1,036 billion gallons for combined "boiler feed and sanitary services" (Bureau of the Census 1971)[5] the intake for steam generation alone in 1968 is assumed to have been approximately 1,000 billion gallons.

Recycling condensed steam back to the boiler will vary from zero for some industrial uses and district steam generating plants to almost 100 per cent for thermal power generation plants.

Boiler makeup will vary from negligible losses and blowdown in the thermal power plants to substantially the total water intake in district steam generating plants with no return of steam condensate. Even for these district steam generating plants, the condensate usually goes to a sewer from which it ultimately returns to a surface water course and so cannot be said to have been consumed. It is estimated that 10 per cent of the intake water used for boiler feed in industrial plants is either lost to the atmosphere or incorporated in products. Thus, the total water consumption for steam generation is about 100 bgy.

Discharge is boiler blowdown and steam condensate that is lost to sewers. This corresponds to the difference between intake and consumption or 900 bgy (Bureau of the Census 1971).[5]

Cooling Waters Once-through cooling water use during 1968 in industry other than commercial power generation was at the rate of approximately 3,000 bgy for steam electric power generation, and 7,000 bgy for other uses (Bureau of the Census 1971).[5] It is estimated that water recirculation for cooling in these plants was at least 20,000 bgy.

Total cooling water drawn from source by commercial steam electric power plants approximated 58,200 bg in 1970, including the Tennessee Valley Authority and a number of other publicly owned steam electric plants (Federal Power Commission 1971).[6] Recirculating cooling systems in these plants are estimated to provide 10 to 15 per cent of the total cooling requirements for this industry, which represents a small proportion of the total water intake. The use of recirculating cooling water systems is expected to increase rapidly as cooling water volume requirements increase and as restrictions become more stringent on maximum discharge temperatures.

Including sea water, approximately one-third of the water used for once-through cooling was brackish. Some plants recirculate brackish water, but because of the limited number of such operations, water quantities have not been established for this type of cooling.

Recirculating cooling water systems require a much smaller withdrawal for makeup than the amount withdrawn for once-through cooling systems of equivalent heat removal capacity. Although the rate of recirculation is frequently only two or three times as high as the once-through flow rate for equivalent cooling, the withdrawal rate for once-through cooling may be 20 to 80 times as high as that required for makeup to a cooling tower system of equivalent cooling capacity. The actual reduction in volume of water drawn from source by recirculation depends upon the temperature difference across the cooling tower and the chemical composition of the recirculating water. No data are available to provide actual totals of water withdrawn from sources for cooling tower makeup or returned as cooling tower blowdown.

TABLE VI–6—Total Water Quantities Used For Once-Through Cooling

Use	Water quantities (bgy)
Industrial steam-electric generation	3,000
Other	7,000
Commercial power	58,000
TOTAL	68,000

An increasing number of plants use municipal sewage treatment plant effluent or industrial waste treatment plant effluent as makeup water for recirculation through cooling towers. This, in effect, is a double recirculation of available water supplies or, from another viewpoint, an elimination of most water withdrawal from natural sources. The use of such treatment plant effluent as cooling tower makeup must be approached with caution since inadequate removal of organic matter, particularly detergents, nitrogen compounds, and phosphates, in the treatment plant can create severe operating difficulties in cooling towers as a result of foaming, excessive microbiological growths, or calcium phosphate deposits.

Significant Indicators of Water Quality

Table VI-2 shows the quality characteristics of waters that have been treated by existing processes to produce waters acceptable for boiler makeup and cooling. In general terms, the water fed to a steam boiler should be of such quality that it:

- forms no scale or other deposits;
- causes no corrosion of the metals present in the boiler, feedwater system, or condensate return system;
- does not foam or prime;
- does not contain enough silica to form turbine blade deposits in high-pressure boilers.

In order to produce waters meeting these requirements, the waters from available supplies are first processed through external water treatment equipment, such as filters or ion exchangers, and then internal conditioning chemicals are added. Table VI-5 shows quality requirements for boiler feed waters that have already been processed through a required external water treatment equipment, but have not yet received any required application of internal conditioning chemicals.

The values for boiler feed water quality requirements must be considered only as rough guides. Usually, more liberal maximum concentrations are acceptable in feed water for boilers operating at lower pressures within each range. However, even here there are many deviations in practice because of differences in the construction and operation of different boilers. For example, all other things being equal, the higher the makeup rate, the higher the quality of the makeup water should be.

Ideally, cooling waters should be:

- nonscaling with reference to such limited solubility compounds as calcium carbonate, sulfate, and phosphate;
- nonfouling as a result of formation of sedimentary deposits or of biological growths;
- noncorrosive at operating flow rates and skin temperatures to materials of construction in the system, including metals, wood, concrete, asbestos-cement, and plastics.

Table VI-5 shows quality requirements for cooling waters both once-through and makeup for recirculation, subsequent to any required external treatment (other than so-called side stream filters or centrifugal separators for removal of suspended matter from recirculating cooling waters) but prior to the addition of any internal treatment chemicals.

For both steam generation and cooling, the more nearly the composition of water at the source (Table VI-2) approaches the quality required at point of use (Table VI-5), the more desirable it is. However, in some instances it may be preferable to resort to a lower-quality, lower-cost raw water, if economic treatment can be expected to yield a lower overall cost.

Water Treatment Processes

The water treatment processes marked by an X in Table VI-7 are used in producing water of the appropriate quality for either cooling or boiler makeup. In addition to external treatment processes outlined in Figure VI-1, commonly used internal conditioning processes are also included in Table VI-7. Not all of these processes are used for the treatment of any individual intake water. Only those processes to produce the quality required are used.

TABLE VI-7—Processes Used in Treating Water for Cooling or Boiler Makeup

	Cooling		
	Once through	Recirculated	Boiler makeup
Suspended solids and colloids removal:			
Straining	X	X	X
Sedimentation	X	X	X
Coagulation		X	X
Filtration		X	X
Aeration		X	X
Dissolved solids modification Softening			
Cold lime		X	X
Hot lime soda			X
Hot lime zeolite			X
Cation exchange Sodium		X	X
Alkalinity Reduction Cation exchange			
hydrogen		X	X
Cation exchange hydrogen and sodium		X	X
Anion exchange			X
Dissolved solids removal:			
Evaporation			X
Demineralization		X	X
Dissolved gases removal:			
Degasification			
mechanical		X	X
vacuum	X		X
heat			X
Internal conditioning:			
pH adjustment	X	X	X
Hardness sequestering	X	X	X
Hardness precipitation			X
Corrosion inhibition General		X	X
Embrittlement			X
Oxygen reduction			X
Sludge dispersal	X	X	X
Biological control	X	X	

The fact that external water treatment processes may be a source of potential waste water problems has been mentioned. The blowdown from evaporative systems, both boiler waters and recirculated cooling water, can become one of these potential problems. This can be caused by increased concentration of dissolved solids from the evaporative process, by increased suspended solids scrubbed from the air or developed by growth of biological organisms, or by chemicals added to the recirculated water for control of scale, corrosion, or biological growths.

TEXTILE MILL PRODUCTS (SIC 22)

Description of the Industry

In the *1967 Census of Manufacturers* (Bureau of the Census 1971),[5] the textile industry was reported to employ 929,000 individuals in 7,080 plants, adding over $8 billion of value annually through manufacturing. The *Statistical Abstract of the United States: 1969* (U. S. Department of Commerce, Bureau of the Census 1969)[18] reported that the industry invested over $1 billion in new facilities during that year.

Cotton is the most important fiber in American textiles and represents about one-half of the total fiber used. Wool and rayon approximate 10–15 per cent of the consumption, and uses of noncellulosic synthetic fibers are increasing rapidly.

The basic processes involved in finishing textiles include scouring, dyeing and printing, bleaching, and special finishing (U. S. Department of The Interior Federal Water Pollution Control Administration 1968).[21] Wool is usually scoured before being woven into cloth. Cotton is woven in the dry state except for stiffening of the warp, known as sizing. Subsequently, the cloth is scoured to remove size and natural impurities before bleaching and dyeing. Synthetic fibers do not require scouring, but cloth made from blends of synthetics and natural fibers may be scoured before finishing.

Water of proper quantity and quality is essential to the textile industry. Most of the early mills in the United States were located in New England, where rivers were capable of providing water for power and ample high quality process water with only minimum treatment. In recent years the trend has been for textile plants to move to the Southeast and locate closer to the raw material (cotton). Need for water as a source of energy has diminished because of the ability to operate with various fuels and electricity. Raw water quality has become less important, because developments in treatment technology have made it economically possible to produce water of adequate quality with the existing wide range of raw water characteristics. This combination of circumstances makes raw water supply and quality less vital in determining plant location today, although emphasis on treatment to correct deficiencies in raw water quality continues.

Processes Utilizing Water

Total 1967 water intake for textile industries using over 20 million gallons annually (684 plants) was 154 billion gallons, 71 per cent of which was used as process water. Of all water intake by the industry, 51 per cent is derived from company-developed surface supplies, 10 per cent from ground water, 1 per cent brackish, and 38 per cent from public supplies. Gross water use by textile plants totalled 328 bg, 174 bg of which was reused in 353 of the 684 facilities (Bureau of the Census 1971).[5] Trends in new textile technology are toward increased reuse of water.

Cotton and wool finishing plants use 30,000 to 70,000 gallons per 1,000 pounds of cloth. Synthetic finishing mills use considerably less (3,000–29,000 gal/1,000 lb), because lack of natural impurities reduces washing requirements.

Wool usually is scoured by moving it through a two- to six-bowl "train," the first one or two of which contain detergents or soaps, and alkalis at 30–50 C. Subsequent bowls are for rinsing and often may be operated in counterflow pattern to conserve water. Usually scouring solutions are not recycled, although effluent rinse waters may be used to make up scouring baths.

Cotton scouring removes natural impurities, as well as sizes added during conversion of fibers into cloth. Scouring operations in series of tanks ("J" boxes) are carried out under highly alkaline conditions (pH 12) and temperatures of 80–120 C and must be followed by thorough rinsing to remove residual color and other chemicals. Mercerizing cotton has involved a major use of water in many mills, but mercerizing is decreasing with increased adoption of cotton and synthetic blends.

Bleaching cotton is done generally with chlorine, while hydrogen peroxide is used for wool and blends containing synthetic fibers. Chlorine is used under slightly alkaline solution (pH 9) and hydrogen peroxide under acid conditions (pH 2.5–3.0). Rinsing of bleached fiber or cloth requires high quality water.

Dyeing also requires high quality water. Specific requirements and process conditions vary widely depending on types of fibers and characteristics of dyes employed. Cotton generally is dyed at moderately high pH, wool at slightly acidic pH, and synthetics under various conditions dependent upon character of fiber. Dyeing operations constitute major uses of water in the textile industry.

Significant Indicators of Water Quality

The textile industry employs a great variety of raw materials, chemical additives, and manufacturing processes to meet a broad range of finished product specifications. Accordingly, water quality requirements in this industry vary extensively, depending on circumstances attending uses, and no single listing of recommendations could be meaningful for the industry as a whole.

To be desirable for use in the textile industry, water should be low in iron, manganese, and other heavy metals, dissolved solids, turbidity, color, and hardness; it should be free from undesirable biological forms (Nordell 1961,[13] McKee and Wolf 1963)[9]. Although raw water supplies of rather undesirable quality have been employed successfully by textile industries (see Table VI-2) with appropriate treatment to correct deficiencies, it is apparent that the more closely raw water quality approaches requirements at the point of use (Table VI-8), the more desirable that source would be.

Turbidity and color are objectionable in water used in textile industries, because they can cause streaking and staining. Iron and manganese stain or cause other process difficulties at low concentrations. Hardness is objectionable in many operations, especially in scouring where soap curds may be produced, and in processes where deposits of precipitated calcium and magnesium may adhere to the material. In wool processing, all scouring, rinsing, and dyeing operations may require zero hardness water. Zeolite-softened or deionized water may be used for manufacturing synthetic fibers (Nordell 1961).[13] Nitrates and nitrites have been reported as injurious in dyeing of wool and silk (Michel 1942).[10]

In Table VI-8 typical ranges of desirable maximum concentrations of constituents that have been suggested for waters used in textile production are summarized (Mussey 1957,[12] Nordell 1961,[13] McKee and Wolf 1963,[9] Ontario Water Resources Commission 1970[14]). The values relate to water quality at point of use before addition of internal conditioning or manufacturing process chemicals. Although data in Table VI-8 may give general guidelines to water quality requirements in this industry, each plant must be

TABLE VI-8—Quality Requirements of Water at Point of Use by the Textile Industry[a]

Characteristic	Typical maximum ranges
Iron, mg/l Fe	0.0-0.3
Manganese, mg/l Mn	0.01-0.05
Copper, mg/l Cu	0.01-5
Dissolved solids, mg/l	100-200
Suspended matter, mg/l	0-5
Hardness, mg/l as $CaCO_3$	0-50
Color, units	0-5
Turbidity, units	0.3-5
Sulfate, mg/l	100
Chlorides, mg/l	100
Alkalinity, mg/l as $CaCO_3$	50-200
Aluminum oxide, mg/l Al_2O_3	8
Silica, mg/l SiO_2	25
Organic growths	absent

[a] Water quality prior to addition of substances used for internal conditioning.

considered in light of the manufacturing processes and other circumstances specific to that installation.

Water Treatment Processes

Some ground supplies are capable of furnishing large quantities of water having quality consistent with industry requirements. However, in many instances other factors desirable in plant location can make it necessary to use a raw water supply of quality not meeting process requirements. In particular, most surface sources are not capable of supplying water suitable for textile industry uses without treatment.

The *1967 Census of Manufacturers* (Bureau of the Census 1971)[5] indicated that of 154 bg water intake (for plants using over 20 million gallons annually), 89 bg were treated in some fashion. Table VI-9 summarizes the total quantity of water and water treatment method employed by each process for 1971 and the number of establishments employing them.

Another approach employed by many textile industries is to obtain potable water through purchase from public supplies. Although this often provides a satisfactory arrangement, it must be noted that some waters adequate in quality for potable purposes do not meet requirements for some types of textile processing. Also, methods of treatment employed in some public systems may have adverse effects on water quality for use in the textile industry.

TABLE VI-9—Water Treatment Processes Employed by Textile Industrial Establishments in 1971

Type of process	bgy treated	Number of establishments
Aeration	2	16
Coagulation	52	116
Filtration	70	184
Softening	33	209
Ion exchange	9	27
Corrosion control	30	121
pH adjustment	48	132
Settling	33	64
Other	7	45
Total employing treatment		408
No treatment performed		276

Bureau of the Census 1971[5]

The *1967 Census of Manufacturers* reported discharge of 136 bg by the textile industry, leaving 18 bg (12 per cent) evaporation or incorporation into products (Bureau of the Census 1971).[5] Of the 136 bg discharged, 54 bg received some degree of treatment prior to discharge.

LUMBER AND WOOD PRODUCTS (SIC 24)

Description of the Industry and Processes Utilizing Water

The total amount of lumber used for various purposes in the United States has not changed significantly in the past three decades (Landsberg et al. 1963).[7] There have, however, been some important shifts in the end products manufactured by the industry. The use of pulpwood for veneer logs has shown steady increases. Lumber for use in wooden containers has been declining, as has wood used for fuel, although fuel wood still accounts for almost 15 per cent of lumber use.

In recent years, about 40 per cent of wood consumption has been for building purposes and 20 per cent for the manufacture of a variety of wooden and paperboard containers, furniture, and other wood products. Paper products, other than containers, account for about 12 per cent of lumber consumption. The remaining 13 per cent is used in a variety of wood-related products such as charcoal, synthetic fibers, and distillation products.

The wood and lumber products industry is a relatively small water user. Of the 36,795 establishments surveyed in the *1967 Census of Manufacturers* (Bureau of the Census 1971),[5] only 0.5 per cent or a total of 188 reported the use of 20 million gallons of water or more in 1968. Total water withdrawn by plants using 20 million gallons or more per year showed a decrease from 151 billion gallons in 1964 to 118 billion gallons in 1968. Less than 10 per cent of the water withdrawn by these larger water using plants is given any form of treatment prior to use.

In general, the lumber industry collects logs from the forest and prepares them for use by sawing the logs into various shapes. Earlier in this country's history, logs were cut in the winter when the snow was on the ground to facilitate their transfer by dragging them overland to rivers. The rivers transported the logs to millsites. The logs were frequently left in the water, if they could be fenced off or driven into a backwater to prevent them from going further downstream. While the log was floating, the water prevented it from drying and cracking at the cut end.

Today, lumber may be transported to a mill that may not be near a river. If the logs accumulate, the ends are moistened by floating them in a pond or by spraying the log pile to prevent cracking. The log is frequently debarked by water jets before it is cut into the desired shape.

TABLE VI-10—Quality Characteristics of Waters That Have Been Used by the Lumber Industry

Characteristic	Value
Suspended Solids	3 mm. diameter
pH, units	5 to 9

ASTM 1970[4] or Standard Methods 1971[14]

Some lumber is treated with chemicals to reduce fire hazards, retard insect invasion, or prevent dry rot. These preservative processes use small volumes of water in a preparation of chromates, cupric ions, aluminum ions, silicates, fluorides, arsenates, and pentachlorophenates. Some forest products are processed mechanically or chemically to make a variety of consumer products.

Significant Indicators of Water Quality

There are few significant indicators of water quality for the lumber industry. The suspended solids should be less than 3 millimeters in diameter and the pH should preferably be between 5.0 and 9.0 to minimize corrosion of the equipment (Table VI-10). (Water used for transportation does not qualify as process water.)

Water used to prepare solutions for treatment of lumber should be reasonably free of turbidity and precipitating ions. Frequently, because of the highly toxic nature of these solutions, efforts are made to recycle as much solution as possible. Thus, makeup water is required to compensate for the portion of the solution lost when forced into the lumber under pressure, and thus evaporated during seasoning.

Water Treatment Processes

For the lumber production phase only, straining may be required. Clarification may be practiced for water used in lumber preservation, but this would be necessary in only very small volume.

PAPER AND ALLIED PRODUCTS (SIC 26)

Description of the Industry

The United States is the world's largest producer and user of paper and allied products. The industry's net sales in 1970 were over $21 billion with over 52 million tons of product produced (American Paper Institute 1970).[1] The per capita consumption of paper products in 1969 was roughly 560 pounds per person, an increase of more than 100 pounds per person in the past decade. It is anticipated that close to 62 million tons of paper and paperboard will be produced in the United States in 1980, as compared with 44 million tons in 1965 (Miller Freeman Publications undated).[11]

TABLE VI-11—Basic Categories of the Pulp and Paper Industry

Type of plant	Number of plants in United States 1969
Paper and paperboard	493
Pulp mills	48
Integrated pulp and paper mills	228
Roofing paper mills	77
Converting plants (units owned by pulp and paper companies)	787
Headquarters, offices, research and engineering labs (separate from mills)	152
Totals	1,785

The pulp and paper industries described encompass a number of basic manufacturing processes involved in the production of a wide variety of paperboard and paper products. These include packaging, building materials, and paper products ranging from newsprint to coated and uncoated writing papers, tissues, and a number of other special types of paper and paperboard for domestic and industrial purposes. Table VI-11 shows the basic categories of the industry.

Processes Utilizing Water

The manufacture of pulp and paper is highly dependent upon an abundant supply of water. The major process water uses are for preparation of cooking and bleaching chemicals, washing, transportation of the pulp fibers to the next processing step, and formation of the pulp into the dry product.

The industries involved in the manufacture of paper and allied products rank third in the withdrawal of water for manufacturing purposes (behind primary metal industries and chemical and allied products). Of the 5,890 plants surveyed by the *1967 Census of Manufacturers* (Bureau of the Census 1971),[5] 619 plants reported withdrawing 20 million gallons of water or more in 1968. Table VI-12 shows the amount of water withdrawn in 1964 and 1968 for those plants using more than 20 million gallons per year. More than half of the water withdrawn in 1968 was treated prior to use and recirculated about three times before discharge. Less than 10 per cent of the water withdrawn was consumed in the manufacturing processes.

TABLE VI-12—Total Water Intake and Use—Paper and Allied Products (billion gallons)

Water intake	1968	1964
Total	2,252	2,064
Treated prior to use	1,311	987
Gross water used (Includes recirculated water)	6,522	5,491
Water discharged	**1968**	**1964**
Total	2,078	1,942

Bureau of the Census 1971[5]

TABLE VI-13—Water Process Used by Paper and Allied Products Manufacturing

Manufacturing process	Typical water use in 1,000 gallons/ton product[a]
Wood Preparation	
Hydraulic barking	3
Drum barking	0.3
Wood washing	0.2
Groundwood Pulp	
Stone groundwood	5
Refiner groundwood	5
Cold soda pulp	3
Neutral Sulfite Semichemical	
No recovery	15
With recovery	10
Kraft and Soda Pulping	25
Prehydrolysis	2
Kraft Bleaching	
Semibleach	25
Highbleach	25
Dissolving grades (soft wood)	50
Dissolving grades (hard wood)	50
Acid Sulfite Pulping	
No recovery	70
MgO recovery	9
NH_3 recovery	8
Sulfite Pulp Bleaching	
Paper grade	20
Dissolving grade	45
De-inking Pulp	
Magazine & ledger	28
News	28
Paper Making	
Coarse paper	10
Fine paper	30
Book paper	10
Tissue paper	30
Specialties[b]	
Waste Paperboard	10
Building Products	
Building papers	10
Felts	3
Insulating board	15
Hardboard	13
Exploded	1

[a] Figures shown represent averages over two-week period with 90 percent frequency.

[b] Varies widely depending upon product.

Environmental Protection Agency, unpublished data

Approximately 70 per cent of the water used in the industry was withdrawn from surface supplies. Other water sources were ground water supplies (about 17 per cent) and public water supplies (about 11 per cent). Tidewater accounted for the remainder of the water used. Water withdrawn for process purposes constituted the largest percentage of water used by the industry (about 65 per cent) while the other major water uses were for cooling purposes.

While the industry has been aptly categorized in general terms by SIC code numbers, a typical plant falling under an SIC code may be engaged in a variety of individual manufacturing processes. For this reason, a clearer picture may be obtained by describing water use in terms of manufacturing processes rather than by SIC subcategories. Table VI-13 classifies the processes used in producing pulp and paper products manufactured in the United States.

These processes have been categorized based on the logical sequence in production along with the use of water made by each process. Presenting the information in this fashion makes it possible to estimate water requirements for any individual mill based on the manufacturing processes employed and the tons of product produced.

Significant Indicators of Water Quality

A survey by the Technical Association of the Pulp and Paper Industry (TAPPI Water Supply and Treatment Committee *unpublished data* 1970,[27] Walter 1971)[23] of water quality requirements for the paper industry revealed a total of 23 specific water quality problems resulting from impurities in the raw water source. The primary causes of the problems centered on hardness, alkalinity, turbidity, color, and iron. In addition, manganese along with iron and color was reported as having an adverse effect on bleaching processes; manganese also produced black spots on paper. In some cases, algae and bacteria interfered with the paper machine operations by causing slime. In addition to causing scale in the mill water supply, high hardness interferes with washing operations and causes fouling in resin sizing and digesting processes. Suspended matter and turbidity interfere with the brightness of the product and cause difficulties by clogging wires and felts in the paper machines. Highly colored waters have an adverse effect on paper brightness and are particularly undesirable for white and dyed papers as well as pulps. Control of pH of the water supply at the mill is important to avoid corrosion of the equipment and for effective use of fillers, sizes, and dyes in the process water.

To avoid some of the problems mentioned above, the *1967 Census of Manufacturers* reported that in 1968 more than one half of the water withdrawn for use by plants in the pulp and paper industry utilizing more than 20 million gallons per day was treated prior to use (Bureau of the Census 1971).[5] The treatment consisted of the various processes shown in Table VI-14.

The source of water and its composition vary widely depending on plant location. The treatment of the mill water supply consequently varies. In general, however, TAPPI

TABLE VI-14—Water Treatment Processes—Paper and Allied Products

Process	Billion gallons treated	Number of establishments
Aeration	62.8	28
Coagulation	821.8	194
Filtration	890.4	272
Softening	116.1	239
Ion exchange	53.5	148
Corrosion control	187.5	126
pH adjustment	357.5	119
Settling	494.5	107
Other	93.1	45
Total	1,311.4	466

Bureau of the Census 1971[5]

TABLE VI-15—Summary of TAPPI Specifications for Chemical Composition of Process Water for Manufacture

Substance—max ppm	Fine paper	Kraft paper		Groundwood papers	Soda and sulfite pulp
		Bleached	Unbleached		
Turbidity (SiO_2)	10	40	100	50	25
Color in platinum units	5	25	100	30	5
Total hardness ($CaCO_3$)	100	100	200	200	100
Calcium hardness ($CaCO_3$)	50	......	......	......	50
Alkalinity to M.O. ($CaCO_3$)	75	75	150	150	75
Iron (Fe)	0.1	0.2	1.0	0.3	0.1
Manganese (Mn)	0.03	0.1	0.5	0.1	0.05
Residual chlorine (Cl_2)	2.0	......	......	......	......
Silica (soluble) (SiO_2)	20	50	100	50	20
Total dissolved solids	200	300	500	500	250
Free carbon dioxide (CO_2)	10	10	10	10	10
Chlorides (Cl)	......	200	200	75	75
Magnesium hardness ($CaCO_3$)	......	......	......	......	50

Technical Association of the Pulp and Paper Industry 1957[17]

indicates that the chemical composition of process water for use by the paper and allied products industry should have the specifications shown in Table VI-15. The production of some specialty papers, however, requires water of considerably higher quality.

CHEMICAL AND ALLIED PRODUCTS (SIC 28)

Description of the Industry

The chemical and allied products industry is quite complex because of its wide range of products and processes. This industry produces more than 10,000 commercial products covering a broad range of uses. Most of the products are converted to another form by other industries before reaching the consumer. Thus, many are little known or understood by the general public.

Processes Utilizing Water

The Bureau of the Census subdivides chemical and allied products into 27 industries. Many of these are shown in Table VI-16, along with estimates of the water intake for process uses by each industry.

Water is essential to most of the processes used in chemical manufacturing. It can be used to separate one chemical from another or to remove a chemical from a gas stream. It can be the medium in which a chemical reaction occurs. It can be employed as a carrier to introduce materials into a reaction system or to dissolve or wash impurities from a product. It often is part of the final product. Water can also be used in the vapor form as steam heat to facilitate chemical reactions or process operations. It can be used in the liquid form to remove heat generated by other chemical reactions or operations. Water is also the product of some chemical reactions.

Generally, the minimum water quality required for a specific process has been determined through experience and is discussed below. In some cases the minimum quality has never been established because the available water in use is acceptable and not necessarily the minimum quality that can be used.

Significant Indicators of Water Quality

The number and diversity of manufacturing facilities in the chemical and allied products industry and their widespread geographical locations in the United States are such that the waters used for process applications vary widely in chemical constituents. Table VI-17 lists some of the quality characteristics in raw water supplies that have been used to provide water for process use in this industry. The figures in Table VI-17 represent extremes, and no water would have all the values shown.

Because of the multitude of products and processes in the chemical industry, only general characteristics can be applied for process water quality required at the point of use. The ranges of quality are so wide, even for similar products, that specific characteristics are not meaningful. In the manufacture of plastic materials and resins, for example, some products require water equivalent to potable water with a maximum total dissolved solids limit of 500 mg/l, while other products require a high level of treatment (i.e., clarification, demineralization, sterilization, and membrane filtration) with a maximum total solids limit well below 1 mg/l.

Low turbidity is the key quality requirement for most of the process water used in the chemical and allied products industries. Other general quality requirements may involve

TABLE VI-16—Process Water Intake by Chemical and Allied Product Industries with Total Water Intake of 20 or More bg During 1968

SIC	Industry group and industry	Process water intake[a]	
		bg	per cent
2612	Alkalies and Chlorine	18.9	2.6
2813	Industrial Gas	5.3	0.7
2815	Cyclic Intermediates and Crudes	19.3	2.6
2816	Inorganic Pigments	21.2	2.9
2818	Organic Chemicals, n.e.c.[b]	394.0	53.7
2819	Inorganic Chemicals, n.e.c.[b]	75.2	10.3
2821	Plastic Materials and Resins	50.9	6.9
2822	Synthetic Rubber	15.1	2.1
2823	Cellulosic Man-made Fibers	30.5	4.2
2824	Organic Fibers, noncellulosic	7.7	1.0
2833	Medicinals and Botanicals	2.7	0.4
2834	Pharmaceutical Preparations	3.9	0.5
2841	Soap and Other Detergents	1.9	0.3
2861	Gum and Wood Chemicals	0.8	0.1
2871	Fertilizers	24.2	3.3
2892	Explosives	28.0	3.8
	Subtotal	699.6	95.4
	Nonlisted Industries	33.8	4.6
28	Chemicals and Allied Products	733.4	100.0

[a] Not including use for sanitary, boiler feed, or cooling water purposes.
[b] Not elsewhere classified.

TABLE VI-17—Quality Characteristics of Waters That Have Been Used by the Chemical and Allied Products Industry

(Unless otherwise indicated, units are mg/1 and values are maximums. No one water will have all the maximum values shown.)

Characteristic	Concentration
Silica (SiO_2)	(*a*)
Iron (Fe)	10
Manganese (Mn)	2
Calcium (Ca)	250
Magnesium (Mg)	100
Ammonia (NH_3)	(*a*)
Bicarbonate (HCO_3)	600
Sulfate (SO)	850
Chloride (Cl)	500
Dissolved Solids	2,500
Suspended Solids	10,000
Hardness ($CaCO_3$)	1,000
Alkalinity ($CaCO_3$)	500
pH, units	5.5-9.0
Color, units	500
Odor threshold number	(*a*)
BOD (O_2)	(*a*)
COD (O_2)	(*a*)
Tempera ure	(*a*)
DO (O_2)	(*a*)

[a] Accepted as received (if meeting other limiting values); has never been a problem at concentrations encountered. ASTM 1970[1] or Standard Methods 1971[16]

total dissolved solids, hardness, alkalinity, iron, and manganese. Where these latter requirements apply, they generally fall in the range of the Drinking Water Standards (U. S. Dept. of Health, Education, and Welfare, Public Health Service 1962).[20] Thus, water from public and private drinking water systems is widely used without further treatment for process applications in the chemical industry. The rigorous water quality requirements for certain products can include nearly all of the characteristics used in describing water quality; however, this high quality represents a very small fraction of the industry's total water use for process purposes.

Table VI-18 shows an example of the quality of process water at point of use in a large chemical plant that manufactures a wide variety of products. The distribution of water processes used is not to be considered typical for the industry. The table is presented to show the levels of treatment applied in merely one multiproduct plant. The process water usage in that plant is 1.2 gallons per pound of product. This is only 2 per cent of the plant's gross water usage; cooling water accounts for all but a slight amount of the balance.

TABLE VI-18—Quality Characteristics of Process Water at Point of Use in a Large Multiproduct Chemical Plant

Treatment process	percent	Dissolved solids mg/l	Hardness (mg/l as $CaCO_3$)
Raw water (screened)[a]	71	95	50
Clarification, filtration, and chlorination[b]	10	95	50
Softening (ion exchange)[c]	14	95	<0.5
Demineralization (ion exchange)	5	<1	—

[a] Dissolved solids and hardness are actual values at this plant location. In most cases water of higher dissolved solids (500 mg/l max) and higher hardness (250 mg/l max) would be acceptable.

[b] Turbidity less than one unit.

[c] Includes steam and boiler feed water used in processes.

Water Treatment Processes

The normal water purification process for raw surface water supplies usually involves clarification (coagulation, sedimentation, filtration). This may be supplemented by softening, demineralization, and other special treatment processes. However, most of the treatment methods shown in Figure VI-1 could be used.

In many cases waters from public supplies or from private wells are acceptable as received and are used without treatment. This constitutes a large portion of the total process water used in the chemical industry.

Generally, the cost of process water treatment is a small part of the overall cost of manufacturing in the chemical industry because of the modest water quality requirements acceptable for many process uses. By contrast, certain processes require exceedingly high-quality water resulting in water treatment costs that can be more than a significant share of the manufacturing costs.

PETROLEUM REFINING (SIC 2911)

Description of the Industry

The principal use of water in the petroleum industry is in refining. Other operations, such as crude oil production and marketing, rely on water but do not use significant amounts. Some water is used in the exploration branch for drilling wells and some is used in the operation of natural gasoline plants, but the amount is insignificant in relation to that used for the refining process.

Refinery Water Consumption Trends

The *1967 Census of Manufacturers* (Bureau of the Census 1971)[5] indicated a gross water use (including recycle) of 7,290 bg. This represented an 18 per cent increase over the 1964 usage. However, the water intake to refineries reporting both in 1964 and 1967 was indicated to be 1,400 bg.

This stable demand can be attributed to the increased use of air for cooling purposes, resulting from increasingly scarce fresh water. In addition, the growing cost of water quality improvement prior to use and prior to final disposal encourages conservation and reuse. Of those refineries included in the 1967 census report, 91 per cent are reusing water.

The total discharge from these refineries was about 1,210 bg, a 7 per cent decrease from 1964.

About 13 per cent of the total water intake by refineries comes from public water supplies, and the remaining 87 per cent comes from company-owned facilities. The company-owned water supply comes from surface (53 per cent),

TABLE VI-19—Summary of Specific Quality Characteristics of Surface Waters That Have Been Used as Sources for Petroleum Water Supplies

Characteristic	Concentration mg/l
Silica (SiO_2)	85
Iron (Fe)	15
Calcium (Ca)	220
Magnesium (Mg)	85
Sodium and Potassium (Na and K)	230
Ammonia (NH_3)	40
Bicarbonate (HCO_3)	480
Sulfate (SO_4)	900
Chloride (Cl)	1600
Fluoride (F)	1.2
Nitrate (NO_3)	8
Dissolved Solids	3500
Suspended Solids	5000
Hardness ($CaCO_3$)	900
Alkalinity ($CaCO_3$)	500
pH, units	6.0–9.0
Color, units	25
Chemical Oxygen Demand (O_2)	1000
Hydrogen Sulfide (H_2S)	20

ground (9 per cent), and tidewater (38 per cent). The use of ground water is being phased out in many locations in favor of impounded surface water. The quality characteristics of surface waters treated to produce waters acceptable for process use are given in Table VI-19.

Processes Utilizing Water

Of the total water intake to all refineries, 86 per cent is used for heat removal by either once-through or recirculating cooling systems, 7 per cent is used for steam generation and sanitary purposes, and 7 per cent for processing. The water distribution in a hypothetical refinery limited to freshwater makeup is shown in Figure VI-2. Here, the distribution is about 56 per cent for cooling, 24 per cent for boilers and sanitary purposes, and 20 per cent for processing. These values differ from the overall average, because the cooling water is circulated.

Process Water Properties

Process water used in refineries may be characterized by the physical and chemical properties of the water. The relevant properties are described in the following paragraphs and in Table VI-20.

A. Inorganic salts that cause deposition and corrosion can be removed from crude oil by a solvent action. Desalting by intimate contact with water is the preferred method. Oil products are frequently purified by washing with acid or caustic solution; diluent water and afterwash water is used in these processes. Catalytic cracking produces quantities of ammonia and carbon dioxide that form deposits unless water is injected into the system to keep them in solution.

B. To transfer heat in numerous operations, barometric condensers are used to create low pressure conditions in fractional distillation. Some catalytic processes require quenching of furnace effluents. Hot water is sometimes pumped through pipelines to facilitate the transfer of high-viscosity petroleum products.

C. Chemical reactions can occur in process water. When quicklime is used in water softening, water enters into the slaking process. At certain times in platforming, water is introduced to chemically condition the catalyst.

D. Water used merely as a carrier must be considered, such as in the periodic cleaning of the plant or in transporting solids through pipelines.

E. Kinetic energy in the form of hydraulically operated cutters is used in decoking furnaces and descaling boiler tubes. Hydraulically operated brushes are used to clean condenser tubes.

F. Some processes use more than one of these properties simultaneously; e.g., water can be introduced into fractionator overhead lines both as a solvent and as a carrier. Ion exchange backwash also relies on these two properties of water.

TABLE VI-20—Process Water Uses in Oil Refineries

Use	Quantity used gal/bbl[a]	Property (see above)	Treatment (see page 387)	Recommendations
Washing	1.5–6.0	D & E	1	Recycled plant effluent is satisfactory.
Desalting	2.0–8.0	A	2	Precipitation of calcium and magnesium salts are undesirable in this process.
Barometric condenser	3.0–6.0	B	1	Recycled plant effluent may be satisfactory. Caution should be exercised because components in the effluent can react with components in the gaseous material being condensed. These reactions, occurring in intimate contact with water, can result in the formation of stable emulsions and/or calcium soaps, which would require downstream chemical treatment.
Caustic dilutant	0.1–0.5	A	2	Calcium, magnesium, carbonate, and bicarbonates are undesirable.
Absorber injection	0.4–1.5	A	2	Calcium salts are undesirable.
Flue Gas quench	0.5–2.0	B	3	Deionized water or steam condensate must be used in this process.
Water wash after caustic	0.1–0.4	A	2	Calcium and magnesium salts are undesirable.
Tank ballast				Sea water is satisfactory.
Furnace quench		B	2	Recycled steam condensate employed for this process.
Fractionator O.H. injection	0.1–0.3	A & D	3	Deionized water or steam condensate must be employed in this process.
Pipelines		B & C	2	Raw water supply with Ryznar Index adjusted below 6.0.
Lime slaking	3.0–7.0	C	1	Raw water supply satisfactory. Recycled plant effluent not satisfactory.
Ion exchange backwash	0.1–0.3	A & D	2	Raw water supply or ion exchanged water, depending upon type of ion exchange.

[a] Gallons of water per barrel of crude oil processed. Refinery capacities are in the range of 20,000 to 180,00 barrels of crude oil per day.

Process Water Treatment

The treatments of refinery process water before use generally fall into three categories. These are shown below and in Table VI-20.

1. No treatment needed. The dissolved and suspended solids are limited only by the restrictions on the plant effluent. In many instances, the plant waste discharge can be recycled.

2. Some treatment, external or internal, needed. Some normal constituents of water undergo physicochemical changes, e.g., calcium carbonate is precipitated by heat. These must be removed or neutralized.

3. Complete of removal solids needed. Usually, these

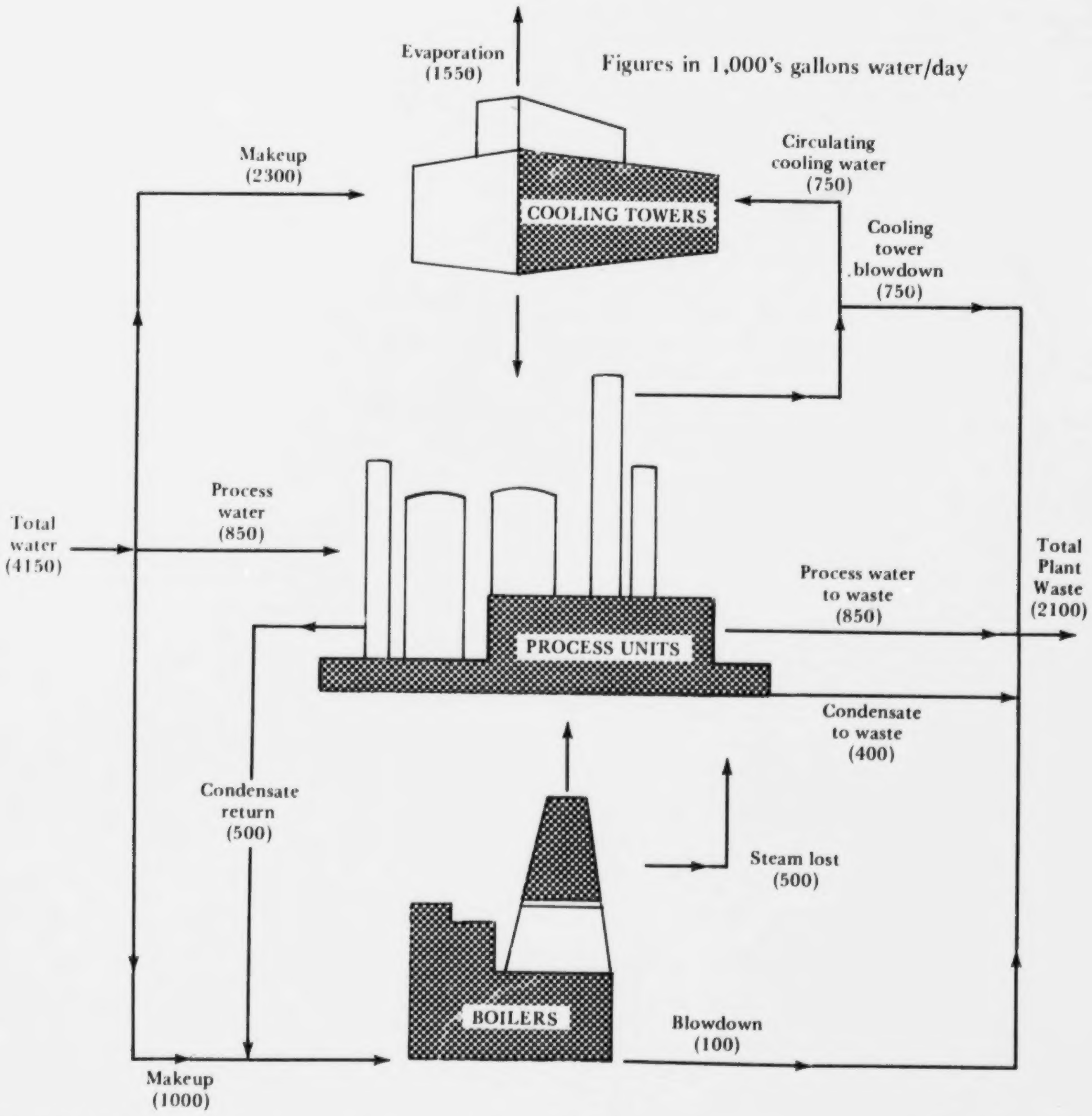
Evaporation (1550)
Figures in 1,000's gallons water/day
Makeup (2300)
Circulating cooling water (750)
COOLING TOWERS
Cooling tower blowdown (750)
Total water (4150)
Process water (850)
Process water to waste (850)
Total Plant Waste (2100)
PROCESS UNITS
Condensate to waste (400)
Condensate return (500)
Steam lost (500)
BOILERS
Blowdown (100)
Makeup (1000)

FIGURE VI-2—Water Distribution in a Hypothetical $55 Million Refinery That Processes 50,000 bbl./Day of Crude (Courtesy of Chemical Engineering Magazine)

waters are vaporized and any water soluble salts remaining are undesirable. These waters may be deionized water or steam condensate.

PRIMARY METALS INDUSTRIES (SIC 33)

Description of the Industry

The primary metals industrial group is defined in the SIC Manual as those "establishments engaged in the smelting and refining of ferrous and nonferrous metals from ore, pig, or scrap; in the rolling, drawing, and alloying of ferrous and nonferrous metals; in the manufacture of castings, forgings, and other basic products of ferrous and nonferrous metals; and in the manufacture of nails, spikes, and insulated wire and cable. The major group also includes the production of coke." (U. S. Executive Office of the President, Bureau of the Budget 1967).[22]

Process water utilization by the primary metals industry as given in the *1967 Census of Manufacturers* (Bureau of the Census 1971)[5] is summarized in Table VI-21. The production of iron and steel utilized almost 88 per cent of all process water used by the industry. For this reason, water quality requirements have been included only for this segment of the industry.

Processes Utilizing Water

The iron and steel industry as defined for this report includes pig iron production, coke production, steel making, rolling operations, and those finishing operations common to steel mills, such as coke reduction, tin plating, and galvanizing. Although many steel companies operate mines for ore and coal, this Section does not dismiss ore beneficiation plants, coal cleaning plants, or fabricating plants for a variety of specialty steel products.

Most of the iron and steel making facilities in the United States are centered in integrated plants. These have generally been located in the Midwest and East where major water sources are available. A few mills have been built in water-short areas because of economic advantages that outweighed the increased cost of recirculating water. The major processes involved in the manufacture of steel require process water, some in several ways. The succeeding paragraphs present a brief description of the process and the process use of water.

TABLE VI-21—Process Water Utilization

Industry	SIC No.	Process water used, 1968 bg.
Iron and steel production	331	1,049
Iron and steel foundries	332	12
Copper industry	3331; 3351	50
Aluminum industry	3334; 3352	36
All other primary metal industries		60
Total process water, primary metals	33	1,207

Bureau of the Census 1971[5]

The production of coke involves the heating of coal in the absence of air to rid the coal of tar and other volatile products. Process water is used in the direct cooling of the incandescent coke after removal from the coke oven in a process called coke quenching. This quenching process is nothing more than dousing the coke with copious amounts of water.

Pig iron production is accomplished in the blast furnace. Process water is used to cool or quench the slag when it is removed from the furnace. The major use of process water in the blast furnace is for gas cleaning in wet scrubbers. Steel is manufactured in open hearth or basic oxygen furnaces. Process water may be used in gas cleaners for either of these furnaces.

The major products of the steel making processes are ingots. Ingots, after temperature conditioning, are rolled into blooms, slabs, or billets depending upon the final product desired. These shapes are referred to as semifinished steel. Water is used for cooling and lubricating the rolls. These semifinished products are used in finishing mills to produce a variety of products such as plates, rails, structural shapes, bars, wire, tubes, and hot strip. Hot strip is a major product, and the manufacturing process for this item will be briefly described.

The continuous hot strip mill receives temperature conditioned slabs from reheating furnaces. Oxide scale is loosened from the slabs by mechanical action and removed by high pressure jets of water prior to a rough rolling stand, which produces a section that can be further reduced by the finishing stand of rollers. A second scale breaker and series of high pressure water sprays precede this stand of rolls in which final size reductions are made. Cooling water is used after rolling for cooling the strip prior to coiling. Most hot-rolled strip is pickled by passing the strip through solutions of mineral acids and inhibitors. The strip is then rinsed with water.

Much hot-rolled strip is further reduced in thickness into cold rolls in which the heat generated by working the metal is dissipated by water sprays. Palm oil or synthetic oils are added to the water for lubrication. After cold reduction, the strip is often cleaned by using an alkaline wash and rinse.

Tin plate is made from cold-rolled strip by either an electrolytic or hot-dip process, more commonly by the former. The electrolytic process consists of cleaning the strip using alkaline cleaners, rinsing with water, light pickling, rinsing, plating, rinsing, heat treating, cooling with water (quenching), drying, and coating with oil. The galvanizing or coating of steel strip with various other products is carried out basically by the same general scheme as tinning.

The volume of water used in the manufacture of steel is a variable that depends on the quantity and quality of the available water supply. The quantity presently being used varies from a minimum of about 1,500 gal/ton of product, where water is reused intensively, to about 65,000 gal/ton, where water is used on only a once-through basis. Both of these figures include total water utilized, not just process water. These figures contrast the range of water intake between plants in areas having extremely limited water supplies and those in areas with almost unlimited water supplies.

Data on the amount of process water required as compared with other water uses indicate that only 24 per cent of the water taken into a steel plant is termed process water (Bureau of the Census 1971).[5] Representatives of the industry have indicated that process water may account for as much as 30 to 40 per cent of the total water intake.

Recycling of water is receiving much attention from the industry as a method to reduce water utilization, reduce stream pollution, and minimize the cost of controlling this pollution. Although individual plants within the iron and steel industry have been practicing reuse of water to varying degrees for some years, the major changes are yet to come. According to the *1967 Census of Manufacturers* (Bureau of the Census 1971),[5] the gross water used in the iron and steel industry (SIC 331) in 1968 was approximately 6,500 billion gallons. This gross water use when compared with a water intake of about 4,400 billion gallons indicates that 2,100 billion gallons were reused. This quantity reflects total water reuse, not just of process water. The consumption of water by the industry amounted to approximately 263 billion gallons in 1968. (No corresponding calculation can be made because no data on process water discharge are available.)

Significant Indicators of Water Quality

The quality of surface waters that are being utilized by the iron and steel industry varies considerably from plant to plant. The desired quality of water for various process uses is difficult to define. For a few processes using relatively small quantities of water, limits on some constituents are known. For most of the process water used, however, only a few of the water quality characteristics have been recognized as a cause of operational problems. For the other characteristics or properties neither the technological nor economical limits are known. (However, the quality of the water available has been much less important than the quantity in determining where a steel mill should be built.) Ranges of values for the selected quality characteristics for existing supplies are listed in Table VI-22. The water quality indicators that are considered important to the industry are settleable, suspended, and dissolved solids; acidity and alkalinity; hardness; pH; chlorides; dissolved oxygen; temperature; oil; and floating materials.

TABLE 22—*Quality Requirements of Water at Point of Use for the Iron and Steel Industry (SIC 33)*

(Unless otherwise indicated, units are mg/l and values that normally should not be exceeded. Table indicates quality of the water prior to the addition of substances used for internal conditioning.)

Characteristics	Quenching, hot rolling, gas cleaning	Cold rolling	Selected rinse waters	
			Partially Softened	Demineralized
Settleable solids	100	5.0	5.0	0.1
Suspended solids	(*a*)	10	5.0	0.1
Dissolved solids	(*a*)	(*a*)	(*a*)	0.5
Alkalinity ($CaCO_3$)	(*b*)	(*b*)	(*b*)	0.5
Hardness ($CaCO_3$)	(*b*)	(*b*)	100	0.1
pH, units	5-9	5-9	6-9	(*d*)
Chloride (Cl)	(*a*)	(*a*)	(*a*)	0.1
Dissolved Oxygen (O_2)	(*c*)	(*c*)	(*c*)	(*c*)
Temperature, F	100	100	100	100
Oil	(*a*)	1.0	1.0	0.02

[a] Accepted as received if meeting other limiting values: has never been a problem at concentrations encountered.
[b] Controlled by treatment for other constituents.
[c] Minimum to maintain aerobic conditions.
[d] Concentration not known.
ASTM 1970[4] or Standard Methods 1971[16]

Water Treatment Processes

Most integrated steel plants have two or more process water systems. One system is the general plant water supply. It receives only mechanical skimming and straining for control of floating and suspended materials that could harm pumps and possibly internal conditioning. This water is used for such diverse tasks as coke quenching, slag quenching, gas cleaning, and in the hot-rolling operations. For some of these operations, many mills use effluent from another process or recycle water in the same process, and the water might actually be of very poor quality. However, the only limits for these process uses which could be established based on present knowledge are those listed in Table VI-22. The other process waters used by the steel industry comprise only 2 to 5 per cent of the total volume but often require considerably improved quality.

Almost universally, one of these two improved supplies is clarified while the second is, in addition, either softened or demineralized. The clarified water is usually a coagulated, settled, and filtered supply that is either treated by the steel company or purchased from a municipality. The use for this water is mainly in the cold-rolling or reduction mill where surface properties of the product are particularly important.

The softened or demineralized water is required for rinse waters following some pickling and cleansing operations. The more particular processes from a water quality point of view are the coating operations, such as tin plating, galvanizing, and organic coating. Some plants use softened and others demineralized water for identical purposes. The quality limits desired for these two types of water, softened and demineralized, are given in Table VI-22.

FOOD CANNING INDUSTRY (SIC 2032 AND 2033)

Description of the Industry

The U. S. canning industry is comprised of about 1,700 canneries. These plants produce some 1,400 canned food items such as fruits, vegetables, juices, juice drinks, seafoods, meats, soups, and specialty products. In 1970, canned foods amounted to about 28 billion pounds packed in 938 million standard cases. The quantities of the major products are: vegetables, 294 million cases; fruits, 153 million cases; juices, 130 million cases; fish, 26 million cases.

Processes Utilizing Water

One of the most important operations in commercial canning is thorough cleaning of the raw foods. The procedures of cleaning vary with the nature of the food, but all raw foods must be freed of adhering soil, dried juices, insects, and chemical residues. This is accomplished by subjecting the raw foods to high-pressure water sprays while being conveyed on moving belts or passed through revolving screens. The wash water may be fresh or reclaimed from an in-plant operation, but it must contain no chemicals or other materials in concentrations that adversely affect the quality or wholesomeness of the food product.

Washed raw products are transported to and from the various operations by means of belts, flumes, and pumping systems. These involve major uses of water. Although the freshwater makeup must be of potable quality, recirculation is practiced to reduce water intake. Chlorination is used to maintain recycled waters in a sanitary condition.

Another major use of water is for rinsing chemically peeled fruits and vegetables to remove excess peel and caustic residue. Water of potable quality must be used in the final rinsing operation.

Green vegetables are immersed in hot water, exposed to live steam or other sources of heat to inactivate enzymes and to wilt leafy vegetables, thus facilitating their filling into cans or jars. Blanching waters are recirculated, but makeup waters must be of potable quality. Steam generation, representing about 15 per cent of water intake, when used for blancing or injection into the product must be produced from potable waters free of volatile or toxic compounds. Syrup, brine, or water used as a packing medium must be of high quality and free of chlorine.

After heat processing, the cans or jars are cooled with large volumes of water. This water must be chlorinated to prevent spoilage of the canned foods by microorganisms in case cooling water is aspirated during formation of a vacuum in the can.

Figure VI-3 shows a flow sheet of the various uses of water and origin of waste streams.

Most fruit and vegetable canning, as opposed to canning of specialty products, is highly seasonal. The demand for water may vary 100-fold throughout the months of the year. The water-demand variation may be severalfold even for plants that pack substantial quantities of nonseasonal items.

The gross quantities of water used per ton of product vary widely among products, among canneries, and during years in the same cannery. The proportion of gross water supplied by recirculation has increased over the years, and

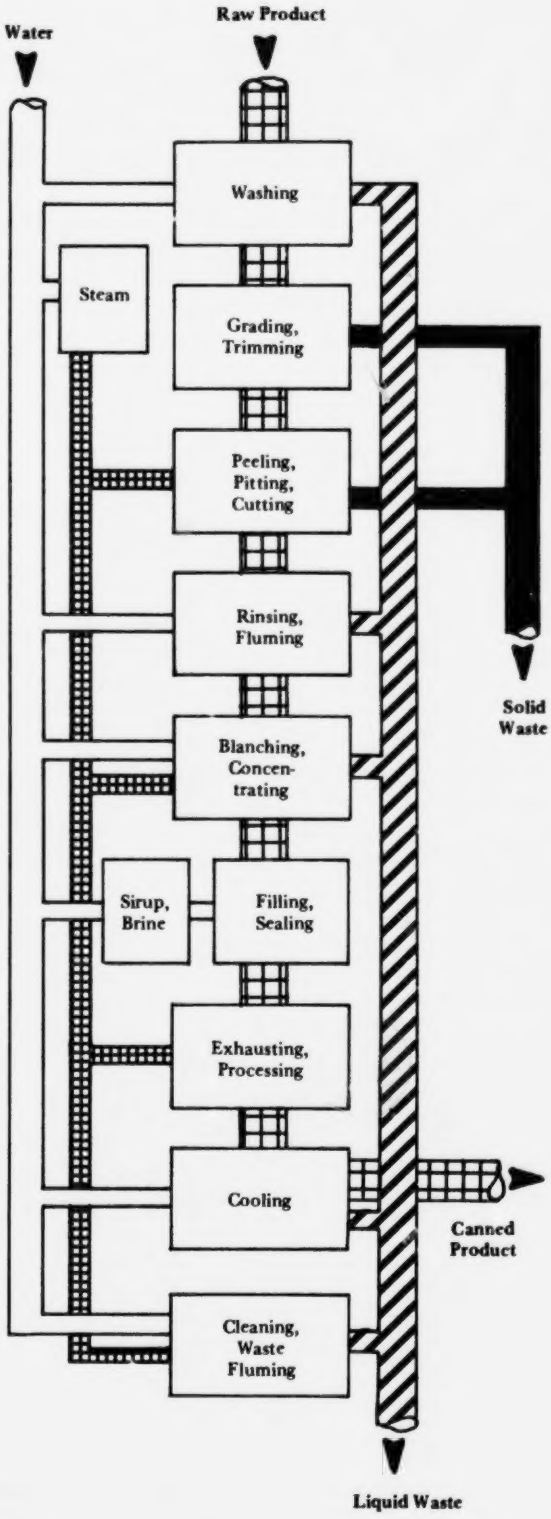

FIGURE VI-3—Uses of Water in Food Canning Industry

TABLE VI-23—Gross Water Intake (annual use over 20 mg) for Canning Plants

Item	Water quantity (bgy)	Percent of intake quantity
Intake	59	100
Reuse	35	59
Consumption	6	10
Discharge	53	90

the trend is expected to continue. A tendency has been noted to use more water per ton of product as the proportion of recirculated water increases. New methods of processing are being evaluated that will reduce the amount of water being used for a given operation and will discharge less organic matter into the wastewater. The trend toward more recirculation of water will continue to increase. As recirculation increases, methods will be employed to improve the quality of the recirculated water and to reduce the amount of fresh water added to the system. Unfortunately, the maximum use of reclaimed water is hindered by specific federal and state regulations originally adopted for other guiding principles that do not now necessarily apply.

The same problem occurs with water conservation, whereby regulations in certain instances demand fixed volumes of water use that, because of process and equipment changes, are no longer necessary.

Table VI-23, gives the rate of gross water intake as based on the *1967 Census of Manufacturers* (Bureau of the Census 1971)[5] for canning plants.

A breakdown of the quantities and percentages of the total water used in the various process operations based on data from the National Canners Association is as follows, Table VI-24.

TABLE VI-24—Total Water Use in Canning Plants

In-plant use	Water quantity (bgy)	Percent of total use
Raw product washing	14.1	15
Product transport[a]	9.4	10
Product preparation[b]	9.4	10
Incorporation in product[c]	5.6	6
Steam and water sterilization of containers	14.1	15
Container cooling	33.9	36
Plant cleanup	7.5	8

[a] Fluming and pumping of raw product.
[b] Blanching, heating, and soaking of product.
[c] Preparation of syrups and brines that enter the container.

Significant Indicators of Water Quality

Of the 48 billion gallons of water intake for canned and cured seafoods and canned fruits and vegetables 24 billion gallons were drawn from public surface water supplies and more than 20 billion gallons from groundwater sources. Approximately 4 billion gallons came from private surface water supplies.

The quality of raw waters for use in the food canning industry should be that prescribed in Section II on Public Water Supplies in this Report.

Table VI-25 has been prepared to indicate the quality characteristics of raw waters that are now being treated for use as process waters in food canning plants. The values given are not intended to imply that better quality waters are not desirable or that poorer quality waters could not be used in specific cases. Significant water quality requirements for water at point of use are given in Table VI-26.

Although the quality characteristics indicated in Table VI-26 may be desirable, it is recognized that many sources of water supplies contain chemicals and other materials in excess of the indicated levels, but with advance treatment these waters may also provide any quality desired at a price.

If the water needs of the nation are projected into the future, the time may come when a completely closed-cycle system will be required in some areas. This means that the waste effluent from a food plant may have to be treated to achieve a high quality water for reuse.

Water Treatment Processes

Surface waters used by the food canning industry require treatment before use as process waters. Usually, this treatment involves coagulation, sedimentation, filtration, and disinfection. More extensive treatment may be required for those waters incorporated in the product.

Container cooling waters are routinely treated by heavy chlorination to render them free of significant types of bac-

TABLE VI-25—Quality Characteristics of Surface Waters That Have Been Used by the Food Canning Industry

(Unless otherwise indicated, units are mg/1 and values are maximums.)

Characteristic	Concentration mg/l
Alkalinity ($CaCO_3$)	300
pH, units	8.5
Hardness ($CaCO_3$)	310
Calcium (Ca)	120
Chlorides (Cl)	300
Sulfates (SO_4)	250
Iron (Fe)	0.4
Manganese (Mn)	0.2
Silica (SiO_2) dissolved	50
Phenols	(*a*)
Nitrate (NO_3)	45
Nitrite (NO_2)	(*c*)
Fluoride (F)	(*a*)
Organics: carbon chloroform extract	0.3
Chemical oxygen demand (O_2)	(*b*)
Odor, threshold number	(*a*)
Taste, threshold number	(*a*)
Color, units	5
Dissolved solids	550
Suspended solids	12
Coliform, count/100 ml	(*a*)

[a] As specified in Water Quality Recommendations for Public Water Supply in this Report.
[b] Accepted as received (if meeting other limiting values); has never been a problem at concentrations encountered
[c] Not detectable by test.
ASTM 1970[4] or Standard Methods 1971.[24]

TABLE VI-26—*Quality Requirements of Water at Point of Use by the Canned, Dried, and Frozen Fruits and Vegetables Industry*

(Unless otherwise indicated, units are mg/l and values that normally should not be exceeded. The Table indicates quality of water prior to the addition of substances used for internal conditioning.)

Characteristic	Canned specialties (SIC 2032) Canned fruits and vegetables (SIC 2033) Dried fruits and vegetables (SIC 2032) Frozen fruits and vegetables (SIC 2037) mg/l
Acidity (H_2SO_4)	0
Alkalinity ($CaCO_3$)	250
pH, units	6.5–8.5
Hardness ($CaCO_3$)	250
Calcium (Ca)	100
Chlorides (Cl)	250
Sulfates (SO_4)	250
Iron (Fe)	0.2
Manganese (Mn)	0.2
Chlorine (Cl)	(*a*)
Fluorides (F)	1 (*b*)
Silica (SiO_2)	50
Phenols	(3, 4)
Nitrates (NO_3)	10 (*b*)
Nitrites (NO_2)	(*c*)
Organics:	
Carbon tetrachloride extractables	0.2 (*e*)
Odor, threshold number	(*c*)
Taste, threshold number	(*c*)
Turbidity	(*f*)
Color, units	5
Dissolved solids	500
Suspended solids	10
Coliform, count/100 ml	(*f*)
Total bacteria, count/100 ml	(*g*)

[a] Process waters for food canning are purposely chlorinated to a selected, uniform level. An unchlorinated supply must be available for preparation of canning syrups.

[b] Waters used in the processing and formulation of foods for babies should be low in fluorides concentration. Because high nitrate intake is alleged to be involved in infant illnesses, the concentration of nitrates in waters used for processing baby foods should be low.

[c] Not detectable by test.

[d] Because chlorination of food processing waters is a desirable and widespread practice, the phenol content of intake waters must be considered. Phenol and chlorine in water can react to form chlorophenol, which even in trace amounts can impart a medicinal off-flavor to foods.

[e] Maximum permissible concentration may be lower depending on type of substance and its effect on odor and taste.

[f] As required by U.S. Department of Health, Education, and Welfare, Public Health Service (1962[20]).

[g] The total bacterial count must be considered as a quality requirement for waters used in certain food processing operations. Other than aesthetic considerations, high bacterial concentration in waters coming in contact with frozen foods may significantly increase the count per gram for the food. Waters used to cool heat-sterilized cans or jars of food must be low in total count for bacteria to prevent serious spoilage due to aspiration of organisms through container seams. Chlorination is widely practiced to assure low bacterial counts on container cooling waters.

ASTM 1970[4] or Standard Methods 1971[16]

teria. Waters used for washing and transporting raw foods are generally chlorinated, particularly if all or a portion of the water is recirculated. In some cases, waters in which vegetables are blanched may require treatment to reduce hardness.

BOTTLED AND CANNED SOFT DRINKS (SIC 2086)

Description of the Industry

Since 1954 there has been a marked reduction in the number of plants producing soft drinks—from 5,469 in 1954 with a production of 1,176,674,000 cases to 3,230 in 1969 with a production of 2,913,110,000 cases (National Soft Drink Association).* It is obvious that numerous small plants have been discontinued as producing units. This trend continues.

Processes Utilizing Water

In the production of soft drinks, water is used not only in the finished product itself but also for washing containers, cleaning production equipment, cooling refrigeration and air compressors, plant clean-up, truck washing, sanitary purposes (restrooms and showers), lawn watering, low-pressure heating boilers, and air conditioning.

Estimates of the total water quantities utilized in the soft drink industry for all purposes are: intake, approximately 18 bgy; recycle, 4 bgy; consumption, 4 bgy; and discharge, 14 bgy (Bureau of the Census 1971).[5]

The figure of 18 bgy intake is based upon production of 2.9 billion cases per year and an average of 6 gallons of water used per case by the 130 largest plants surveyed that represent only 5 per cent of the plants in the industry. (The figure of 6 gallons per case is based on the limited data now available.)

The *1967 Census of Manufacturers* lists the gross water usage in 1968, including recycle, as 9 billion gallons and total water intake as 8 billion gallons (Bureau of the Census 1971).[5] The reuse of water within the industry has for some years increased and is still increasing as the older and smaller plants are replaced by new and larger plants that use recirculating rather than once-through cooling water equipment, modern bottle washers that use less water per case than older equipment and other devices. The increased use of nonreturnable containers in recent years has resulted in lower quantities of water used for bottle washing.

The consumption figure of 4 billion gallons is based upon the water content of the total quantity of beverage produced in 1968.

The discharge figure of 14 billion gallons is the difference between the estimated 18 billion gallons of intake and the 4 billion gallons of product water.

Significant Indicators of Water Quality

Water that is mixed with flavoring materials to produce the final product must be potable. Likewise, potable water is needed for washing fillers, syrup lines, and other product handling equipment. The water used for washing product containers must also be potable. Although other water uses do not require potability, it has not been customary to use nonpotable water for any purpose in a soft drink plant.

The water that becomes a part of the final product must not only be potable, but must also contain no substances that will alter the taste, appearance, or shelf life of the beverage (Table VI-27). Because beverages are made from many

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* A case is defined as 24 bottles each containing 8 ounces of beverage. In the above figures, bottles larger or smaller than 8 ounces have been converted to 8 ounce equivalents.

TABLE VI-27—Quality Requirements of Water at Point of Use by the Soft Drink Industry (SIC 2086)[a]

(Unless otherwise indicated, units are mg/l and values that normally should not be exceeded. The Table indicates the quality of water prior to the addition of substances used for internal conditioning.)

Characteristic	Concentration mg/l
Alkalinity ($CaCO_3$)	85
pH, units	(*b*)
Hardness ($CaCO_3$)	(*b*)
Chlorides (Cl)	500 (*c*)
Sulfates (SO_4)	500 (*c*)
Iron (Fe)	0.3
Manganese (Mn)	0.05
Fluoride (F)	(*d*)
Total dissolved solids	(*b*)
Organics, CCE	0.2 (*e*)
Coliform bacteria	(*d*)
Color, units	5
Taste	(*e*, *f*)
Odor	(*e*, *f*)

[a] The more important parameters are listed. Although not included in the table, all Drinking Water Standards (U.S. Department of Health, Education, and Welfare, Public Health Service 1962)[20] for potability apply.

[b] Controlled by treatment for other constituents.

If present with equivalent quantities of Mg and Ca as sulfates and chlorides, the permissible limit may be somewhat below 500 mg/l.

[d] Not greater than PHS Drinking Water Standards (1962)[20].

[e] In general, public water supplies are coagulated, chlorinated, and filtered through sand and granular activated carbon to insure very low organic content and freedom from taste and odor.

[f] Not detectable by test.

ASTM 1970[4] or Standard Methods 1971[16].

different syrup bases, however, the concentration and type of substances that affect taste, or other characteristics, are not the same for all beverages. For this reason, a single standard cannot apply to all types of soft drinks.

The majority of plants use only water from a public supply. Some use water from private wells. None use water directly from surface sources. The quality characteristics for intake water are essentially the same as requirements for potable water.

Water Treatment Processes

There are few, if any, public water supplies that are suitable as product water without some in-plant processing. Almost 100 per cent of the bottling plants have as minimum treatment sand filtration and activated carbon purification. About 80 per cent of the plants also coagulate and superchlorinate the water preceding sand filtration and carbon purification. When the total alkalinity of the intake water is too high, lime is used to precipitate the alkaline salts.

There are very few bottling plants whose intake water is so highly mineralized that the brackish taste affects soft drinks. Among the reasons are the facts that flavoring components in soft drinks mask the taste of many brackish waters without altering the taste of the drink and that towns with highly mineralized water supplies are either avoided as locations for bottling plants or suitable private supplies are used.

Uniformity of water composition is most desirable. Control of in-plant processing is difficult when the composition of the water varies from day to day. Surface waters that are subject to heavy biological growths or heavy pollution from organic chemicals are also difficult to process.

Except for process water, most public water supplies are suitable for all other usages without external treatment. Occasionally, cation exchangers are used to soften water for bottle washing, cooling, and boiler feed water, but internal conditioning is used in most plants for scale and corrosion control.

TANNING INDUSTRY (SIC 3111)

Description of the Industry

The leather tanning industry is many industries, as each type of leather constitutes a different process. Basically, there are only three or four types of tannage (vegetable, mineral, combination of vegetable-mineral, and synthetics) but many finishing processes.

Processes Utilizing Water

Water is used in all processes of storage, sorting, trimming, soaking, green fleshing, unhairing, neutralizing, pickling, tanning, retanning, fat-liquoring, drying, and finishing of the hides. It is an essential factor in each process. The chemical composition of the water is considered critical in obtaining the desired quality of leather. There is limited reuse of process water in the tanning industry.

Data on water utilization by the leather tanning and finishing industry as reported in the *1967 Census of Manufacturers* (Bureau of the Census 1971)[5] includes 14.8 bgy intake, 3.7 bgy reuse and recirculation, and 0.8 bgy consumption.

TABLE VI-28—Quality Requirements of Water at Point of Use by Leather Tanning and Finishing Industry (SIC 3111)

(Unless otherwise indicated, units are mg/l and values that normally should not be exceeded. Table indicates the quality of water prior to addition of substances used for internal conditioning.)

Characteristic	Tanning processes	General finishing processes	Coloring
Alkalinity ($CaCO_3$)	(*a*)	(*a*)	(*a*)
pH, units	6.0-8.0	6.0-8.0	6.0-8.0
Hardness ($CaCO_3$)	150	(*b*)	(*c*, *d*)
Calcium (Ca)	60	(*b*)	(*c*, *d*)
Chloride (Cl)	250	250	(*e*)
Sulfate (SO_4)	250	250	(*e*)
Iron (Fe)	50	0.3	0.1
Manganese (Mn)	(*e*)	0.2	0.01
Organics: carbon chloroform extract	(*e*)	0.2	(*c*)
Color, units	5	5	5
Coliform bacteria	(*f*)	(*f*)	(*e*)
Turbidity	(*c*)	(*c*)	(*c*)

[a] Accepted as received (if meeting other listed limiting values); has never been a problem at concentrations encountered.

[b] Lime softened.

[c] Not detectable by test.

[d] Demineralized or distilled water.

[e] Concentration not known at which problems occur.

[f] PHS Drinking Water Standards (1962).[20]

ASTM 1970[4] or Standard Methods 1971[16]

Significant Indicators of Water Quality

The chemical composition of the water is important in producing high-quality leather. For some processes, such as the finishing of leather, distilled or demineralized water is best. The microbiological content of the water is equally important, but this can be controlled by use of disinfectants. The quality requirements at point of use are shown in Table VI-28.

Water Treatment Processes

Most tanning and leather product industries are located in urban areas and use public water supplies or ground water. A few tanneries use surface supplies, usually chlorinated. They may also need additional treatment such as clarification, and iron and manganese removal.

A limited volume of water, whether from the public water supply or company-owned systems, may be softened, distilled, or demineralized.

MINING AND CEMENT INDUSTRIES (SIC 10)

Mining

Description of the Industry Industrial usage of the term mining is broad and includes mining operations and quarrying; extraction of minerals, petroleum, and natural gas; well operations and milling (e.g., crushing, screening, washing, froth flotation); and other processing used to render minerals marketable.

Processes Utilizing Water Mining operations are numerous, and many of them involve the use of water. However, the amount of water used is often relatively small, or its use is simply that of providing a suspending medium (as in coal washing) with minimal requirements of water quality. The principal consideration in these operations is that water acidity be relatively low so that corrosion of equipment is kept to a minimum.

On the other hand, a number of the operations involved in this general category require the use of large quantities of water with certain quality requirements relating to impurity, type, and level. These operations are froth flotation, mine dump leaching, and secondary oil recovery. With regard to froth flotation, an operation extensively used to recover valuable minerals from low-grade ores, large tonnages of material are processed each day. For example, in one large plant, 100,000 tons of copper ore per day are treated for recovery of copper sulfide. Generally, flotation is carried out at approximately 25 per cent solids by weight, and freshwater makeup constitutes about 25 per cent of the total water requirement. In such systems, water is normally recycled so that the impurity level of both inorganic and organic constituents builds up with repeated reuse. It is not possible to list maximal limits of impurity levels for such waters, but the levels found in the processing water of one operating plant (i.e., a copper sulfide concentrator) are listed in Table VI-29. Also listed is the analysis of the freshwater makeup that is added to the recycled water. This combination provides the total process water used for this plant.

TABLE VI-29—Analysis of Typical Freshwater Makeup and Process Water for a Copper Sulfide Concentrator

Water type	Constituent (mg/l)							
	H	Ca	M	O	SO_4	Cl	TDS	pH[a]
Freshwater makeup	100	87	104	....	18	8	140	8.0
Process water	1530	1510	415	345	1634	12	2100	11.7

[a] H is total hardness expressed as $CaCO_3$; Ca is total calcium hardness expressed as $CaCO_3$; M is total alkalinity expressed as $CaCO_3$; O is total hydrate expressed as $CaCO_3$; SO_4 is total sulfate; Cl is total chloride; TDS is total dissolved solids.

This fresh water is excellent for flotation. The actual process water used can probably be best described as one bordering on being problematic. The high Ca^{++} concentration together with the high content of hydroxides of heavy metals (column 0) place this water in this category.

Another process that is used extensively in the industry is the leaching of mine waste for recovery of copper. Large quantities of leach solution—approximately 225 million gallons per day—are added to properties located in this country. Most of the properties are located in arid areas, so that water reuse is mandatory. Solutions returned to the mine dumps for leaching have been subjected to treatment for copper recovery by replacement with metallic iron and then to further treatment to set the level of iron in solution. The analysis of a typical leach solution is presented in Table VI-30. Of these species, the amount of ferric ion is perhaps the most critical, in that if the concentration is too high, precipitation of basic iron sulfate occurs within the dump and renders the dump impermeable to solution flow. In this regard it is also important that there be no concentration of suspended solids in such leach solutions as they too render the dump impermeable to flow of solution. As a result, these solutions are filtered prior to introduction to the mine dump.

Secondary oil recovery has assumed great importance in the oil industry. One of the techniques used in recovering oil is water flooding of a formation. With this technique water is pumped into a formation under high pressure, and a mixture of water and oil is then recovered from another well drilled into the formation. Such a process requires

TABLE VI-30—Typical Analysis of Leach Solution in Dump Leaching of Copper

Constituent	Concentration (mg/l)
Al^{+++}	12,000
Mg^{++}	12,000
Fe^{++}	6,000
Fe^{+++}	6,000
$SO_4^{=}$	64,000
pH	3-3.5

careful consideration of a number of factors, including permeability of the rock of which the formation is composed; type and amount of clay in the rock; ionic composition of the connate water; and composition, solids, and bacterial content of the water injected into the formation. If the clay content of the host rock is of a bentonitic nature (i.e., a swelling type clay, which when used with fresh water is not in equilibrium with the ions contained in the connate water), the clay will swell and render the formation impermeable to water flow. An effective means of obviating this is to re-inject the same water, filtered of solids, into the formation. Another means is to keep the salt content of the water high.

Stabilization of the water exiting from the formation must be considered, because gases such as carbon dioxide, sulfur dioxide, and hydrogen sulfide are released from the water. If these gases are not added to the water prior to re-injection into the formation, the water will not be in equilibrium with the connate water, salts, and rock of the formation. Precipitation of compounds may result, and permeability will be altered.

Waters that are conveniently available are used for water injection. In addition to formation and surface waters, sea water is often used. The composition of sea water and a water from a sand formation are listed in Table VI-31.

Anaerobic bacteria are also a problem in water flooding, since they are capable of producing such compounds as hydrogen sulfide in the water. Effective bactericides are available to control this potential problem.

The quantity of water used in water flooding depends on the production of the well involved. A commonly added quantity would be 400 to 500 barrels per day, which is equivalent to 16,800 to 21,000 gallons per day. In view of all of the secondary oil production using this technique, then, extremely large quantities of water are involved. For example, in 1960 approximately 634 million barrels of oil were produced by injection techniques in California, Illinois, Louisiana, Oklahoma, Texas, and Wyoming (Ostroff 1965).[15]

TABLE VI-31—Composition of Sea Water and a Formation Water Expressed as mg/l.

Constituent	Sea water	Marginulina sand (La.)
$CO_3^{=}$		0
HCO_3^{-}	142	281
$SO_4^{=}$	2,560	42
Cl^{-}	18,980	72,782
Ca^{++}	400	2,727
Mg^{++}	1,272	655
$Na^{+} + K^{+}$	10,840	42,000
Fe (total)	0.02	13
Ba^{++}		24
TDS	34,292	118,524
pH		[illegible].5

Ostroff 1965[15]

Cement

The manufacture of cement involves combining limestone with silica sand, alumina, and iron oxide, crushing and grinding this mixture, burning at high temperature, cooling, and regrinding clinker to fine size. If water is used at all, it is used in the initial grinding step. In terms of water consumed, approximately 200 gallons are used per ton of finished cement.

Because of the high temperatures used in the burning process (approximately 2500 F), water quality requirements are minimal. The alkali content of the process water can be a problem, however, if it is present in relatively high concentration, because the alkali oxides are volatilized and condensed on the fine particulate matter produced during the burning process. If the amount of oxide is relatively high, oxide will build up as the fine particulate matter is recycled to the kiln. Alkali oxide may be removed from the fine particulate matter by water leaching, but this practice results in the problem of disposing of water very high in alkali salts. Even if water leaching is not used, the problem of disposing of the oxide-bearing particulate matter also exists.

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Appendix I—RECREATION AND AESTHETICS

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QUANTIFYING AESTHETIC AND RECREATIONAL VALUES ASSOCIATED WITH WATER QUALITY

Provisions of the Wild and Scenic Rivers Act (U. S. Congress 1968),[13] The National Environmental Policy Act (U.S. Congress 1970a),[14] and the Flood Control Act, Section 209 (U.S. Congress 1970b),[15] have added impetus to the need for quantification of aesthetic and recreational values associated with water quality.

Evaluation Techniques

The two techniques necessary to assess total aesthetic and recreational values are (a) monetary benefit evaluations, and (b) nonmonetary benefit evaluations.

Monetary benefit evaluations usually start by determining costs of visiting a site from various distances and adopt a weighted average based on calculations of individual costs to visit a particular site from various zones and the number of visitors from each zone. The representative unit cost is then multiplied by the total number of expected visitors (the demand) to determine the total minimum benefit. (See Hotelling (1949),[5] Trice and Wood (1958),[12] Clawson (1959),[2] and Knetsch (1963).[6]) Another procedure for imputing dollar values to benefits is to presume that benefits are equal to foregone costs of doing the same thing another way. Frankel (1965)[4] showed that the cost of downstream removal of coliforms at a water treatment plant was less than the upstream cost of disinfection at a waste water treatment plant. The conclusion to be drawn was that the benefits of chlorination at the particular waste water treatment plant were not equal to the costs saved downstream, and hence the practice could be discontinued at the waste water treatment plant.

Nonmonetary benefit evaluations attempt to attach quantitative scales in terms of dollars and dimensionless scores to nonmonetary recreational and aesthetic values. These attempts fall into three categories.

1 Waste treatment evaluation techniques Sonnen (1967)[11] devised a scheme of multipliers ranging from 0 to about 10 that, when multiplied by the identifiable monetary benefits of waste treatment, yielded an estimate of intangible benefits. The value of the multipliers was a function of: (a) the downstream users' local, regional, or national scope; (b) the private or public affiliation of the downstream users; (c) the number of people involved in each downstream use; and (d) the relative importance of each constituent in the waste that might influence the enjoyment or use of the water. Only the subfactor for constituent influence was recalculated for each constituent to be partially removed by the alternative treatment processes under consideration. The objective was a benefit–cost analysis of waste treatment alternatives with intangible benefits given quantitative weight. It was shown in a hypothetical stream discharge case that net benefits calculated with monetary benefits alone were maximized by a less complete removal process than was optimal when nonmonetary benefits were included in the analysis. Partial removals of 27 constituents to serve five downstream users, including recreational and aesthetic use, were evaluated.

Water Resources Engineers, Inc. (1968)[16] modified this procedure to evaluate alternatives for: (1) wastewater reclamation to protect current recreation benefits and to provide more; and (2) protection of a particular water to levels (of coliforms) suitable for harvesting shellfish while other competing uses of the water predominated (WRE 1969).[17] Ralph Stone and Co. (1969),[10] in assessing the value of cleaning up San Diego Bay, asked 27 knowledgeable people to rank the Bay's 12 possible uses, giving weights from 1 to 10 to both the economic value and the social value of each use to the community as the interviewee perceived that value. In both the economic and the social value responses, tourism, fishing, marina activities, and park and recreation use ranked highest while industrial activity rated low, and waste disposal rated last in both responses.

2 Water Resource Project Recreation Evaluation WRE (1970)[18] devised two methods for evaluating intangible benefits as functions of the monetary benefits identified: a "benefits foregone—subjective decision" method, and a "nonmonetary expression of benefits" method. In the former the intangible benefits associated with wild, undeveloped streams are presumed to be equal to the foregone monetary benefits that would accrue to other users if the streams were fully developed. In the latter intangible, aesthetic benefits are presumed to be estimable fractions of the identifiable monetary benefits. These two WRE methods have been demonstrated for both a wild river area and a developed stream in the Pacific Northwest.

3 Ecological Impact Analysis Six notable studies in recent years derived evaluation methods that require ranking sites on various scales, with constant upper and lower limits. (1) Whitman (1968)[19] developed a rating scheme for streams in urban areas based on seven factors related to the environment: three factors are assigned 20 per cent relative weights, and four 10 per cent relative weights. Each stream is to be given a rating from 0 to the upper limit for each factor on the basis of how uniquely each of the subjective criteria is satisfied. (2) Dearinger (1968)[3] developed weighted ratings for subfactors encompassing a range of environmental characteristics including climate, scenery, hydrology, user characteristics, and water quality. (3) Leopold (1969)[7] ranked scenic values by placing each stream in categories that measure the site's uniqueness with respect to all others evaluated. His three major categories embraced physical factors, biological and water quality factors, and human use and interest factors. No superior-inferior ranking was implied for any category.

(4) Morisawa and Murie (1969)[9] presented a 1 to 10 value-rating scale to apply quantitative weight to otherwise subjective stream characteristics, placing major emphasis on total dissolved solids content and sediment load with respect to water quality. (5) Leopold et al. (1971)[8] devised a 3′×3′ score sheet on which 86 "existing characteristics and conditions of the environment" are scored according to how they will be affected by any of 98 possible "actions which may cause environmental impact." Of the 86 characteristics, water quality was only one, although temperature was given a row of its own too. Unfortunately, no explicit score is given to the goodness or badness of the scores, and much subjective decision-making remains after these analyses have contributed what objectivity they can. (6) Battelle-Columbus (1971)[1] desired a hierarchical arrangement of critical environmental quality characteristics arranged in four major categories: ecology, environmental pollution, aesthetics, and human interest. The system measures environmental impacts in environmental quality units (EQU); each analysis produces a total score in EQU based on the magnitude of specific environmental impacts expressed by the relative importance of various quality characteristics as prescribed by a predetermined weighting and ranking scheme.

Current Least-Cost Evaluations

The economic objective for water-quality-oriented projects, such as water and waste treatment plants, has been to meet stipulated water quality standards or criteria at least cost. However, least-cost analysis, which is important and proper at the design stage, has entered water quality management evaluations too soon on most occasions. The hasty assumptions are made that (1) certain uses are to be provided or protected, and (2) water quality criteria to protect those uses are absolutely correct both with respect to constituents named and concentrations assigned. But caveats by the experts throughout this book about lack of scientific evidence to support meaningful criteria attest to the fallacy of these assumptions. Clearly if some prior analysis, such as a benefit-cost analysis including aesthetic values, could demonstrate that secondary treatment of wastes would provide adequate protection of the most justifiable mix of downstream uses in a specific set of circumstances, then least-cost analysis would be the proper tool to determine the cheapest secondary treatment process to install. Unfortunately, the biggest stumbling block to this more nearly ideal sequence of analyses has been the lack of procedures for quantifying all the relevant values discussed above, including both monetary and nonmonetary ones. But it should be recognized that least-cost analysis is properly applied only after the uses to be protected and the quality criteria to protect them have been determined through prior evaluation.

Special Evaluation Problems

There are problems that have not yet been addressed by researchers.

- The perception of median value by the average person enjoying himself or his surroundings has not been normalized. The average recreator is not aware of his environment in terms of the silt load or coliform organism measures that the scientists use to characterize the environment.
- A related problem is that of vicarious pleasure and its benefit to society as a whole.
- There is no method available that defines absolute and relative uniqueness. Methods that rank relative uniqueness on scales of 1 to 10 do not answer the optimal questions of water resource use, and methods like WRE's (1970)[18] cannot claim validity for more that comparative evaluations of projects within a single river basin.
- There is no single, meaningful measure of water quality that can be related to the costs of attaining it and the benefits stemming from it. In his study of waste treatment alternatives, Sonnen (1967)[11] was unsuccessful at separating the benefits that overlapped from removal of one constituent and were undoubtedly counted again in assessing the benefits of removal of others.
- The quantification of aesthetic and recreational values associated with marine and estuarine waters demands particular attention.

Further research must attempt to determine the levels of each constituent that enhance, preserve, reduce, or eliminate use of water. With these quality-use spectra, sociologists, psychologists, economists, engineers, and politicians will eventually be able to characterize objectively the average, normative response of the populace to the environment and to deduce the values and relative values people wish to place on the conditions to be found there.

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Appendix II—FRESHWATER AQUATIC LIFE AND WILDLIFE

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APPENDIX II-A

MIXING ZONES

A. Mathematical Model References

Mathematical models based, in part, on the considerations delineated in General Physical Consideration of Mixing Zones are available for prediction of heated-water discharge from power plants into large lakes (Wada, 1966;[32] Carter, 1969;[6] Edinger and Polk 1969,[10] Sundaram et al. 1969,[30] Csanady, 1970,[7] Motz and Benedict 1970,[22] Pritchard 1971,[27] Stolzenbach and Harleman 1971,[29] Zeller et al. 1971,[36] Policastro and Tokar *in press*),[26] cooling ponds and impoundments (Brady et al. 1969,[4] D'Arezzo and Masch 1970),[8] rivers (Jaske and Spurgeon 1968,[17] Water Resources Engineers 1968,[35] Parker and Krenkel 1969,[25] Kolesar and Sonnichsen 1971),[19] estuaries (Ward and Espey 1971)[33] and ocean outfalls (Baumgartner and Trent 1970).[3]

Mathematical models of the distribution of non-thermal discharges into various receiving systems are also available for diffusion in lakes, reservoirs and oceans (scale effects) (Brooks 1960,[5] Allan Hancock Foundation 1965[1]), diffusion in bays and estuaries where tidal oscillations and density stratification are factors (O'Connor 1965,[23] Masch and Shankar, 1969,[21] Fischer 1970,[12] Leendertse 1970),[20] and dispersion in open channels and rivers (Glover 1964,[13] Bella and Dobbins, 1968,[2] Dresnack and Dobbins, 1968,[9] Fischer 1968,[11] Thackston and Krenkel 1969,[31] Jobson and Sayre 1970,[18] O'Connor and Toro 1970).[24]

Time-of-exposure models are discussed by Pritchard (1971).[27]

B. Development of Integrated Time-Exposure Data For a Hypothetical Field Situation

1. A proposed discharge of a waste containing alkylbenzene sulfonates (ABS) to a lake containing rainbow trout is under consideration. The trout regularly swim parallel to the shoreline where the shallows drop off to deeper water. The expected plume configuration, estimated ABS concentrations, and time of exposure for a swimming trout to various concentrations are shown in Figure II-A-1. No avoidance or attraction behavior is assumed. It is decided that an ET_2 is appropriate for this situation (see Comment a. below).

2. To test if this mixing zone meets necessary water quality characteristics, toxicity bioassays with rainbow trout are performed (see Section III, pp. 118–123). Observe mortality after each exposure to selected concentrations at time intervals of approximate geometric or logarithmic progression: i.e., 10, 15, 30 and 60 minutes; 2, 4, 8, between 12 and 16, 24, and between 30 and 36 hours; 2, 3, 4, and if desired 7, 10 or more days. While only the shorter time periods are involved in this example, greater periods are necessary in some cases. After exposure, trout should be held in uncontaminated water for extended periods so that delayed effects of exposure can be evaluated. While mortality was selected in this example as the response to be assessed, a more conservative physiological or behavioral response would provide a more positive factor of safety.

3. Plot percentage mortality on a probability (probit) scale with time on a logarithmic scale as in Figure II-A-2, and fit by eye a straight line to the set of points for each concentration. The object of this is to determine for each concentration the median lethal time where the fitted line crosses 50 per cent mortality (the ET50) and the ET2, the time causing 2 per cent mortality.

4. Plot the sets of ET50 and ET2 values on logarithmic paper and fit each set of points to create the toxicity curves as in Figure II-A-3.

5. Substitute the information on plume characteristics and time of passage (Figure II-A-1) and the toxicity curves (Figure II-A-3) in the summation of effects formula:

$$\frac{T_1}{\text{ET2 at } C_1} + \frac{T_2}{\text{ET2 at } C_2} + \frac{T_3}{\text{ET3 at } C_3} \cdots + \frac{T_n}{\text{ET2 at } C_n} \leq 1$$

$$\frac{6}{\infty} + \frac{6}{52} + \frac{13}{17} + \frac{12}{52} + \frac{11}{\infty}$$

Since the total is slightly over 1.0 a mortality greater than 2 per cent is expected, and the recommendation is not met. If the total was 1.0 or less, a mortality of 2 per cent or less would be expected and the recommendation would be met.

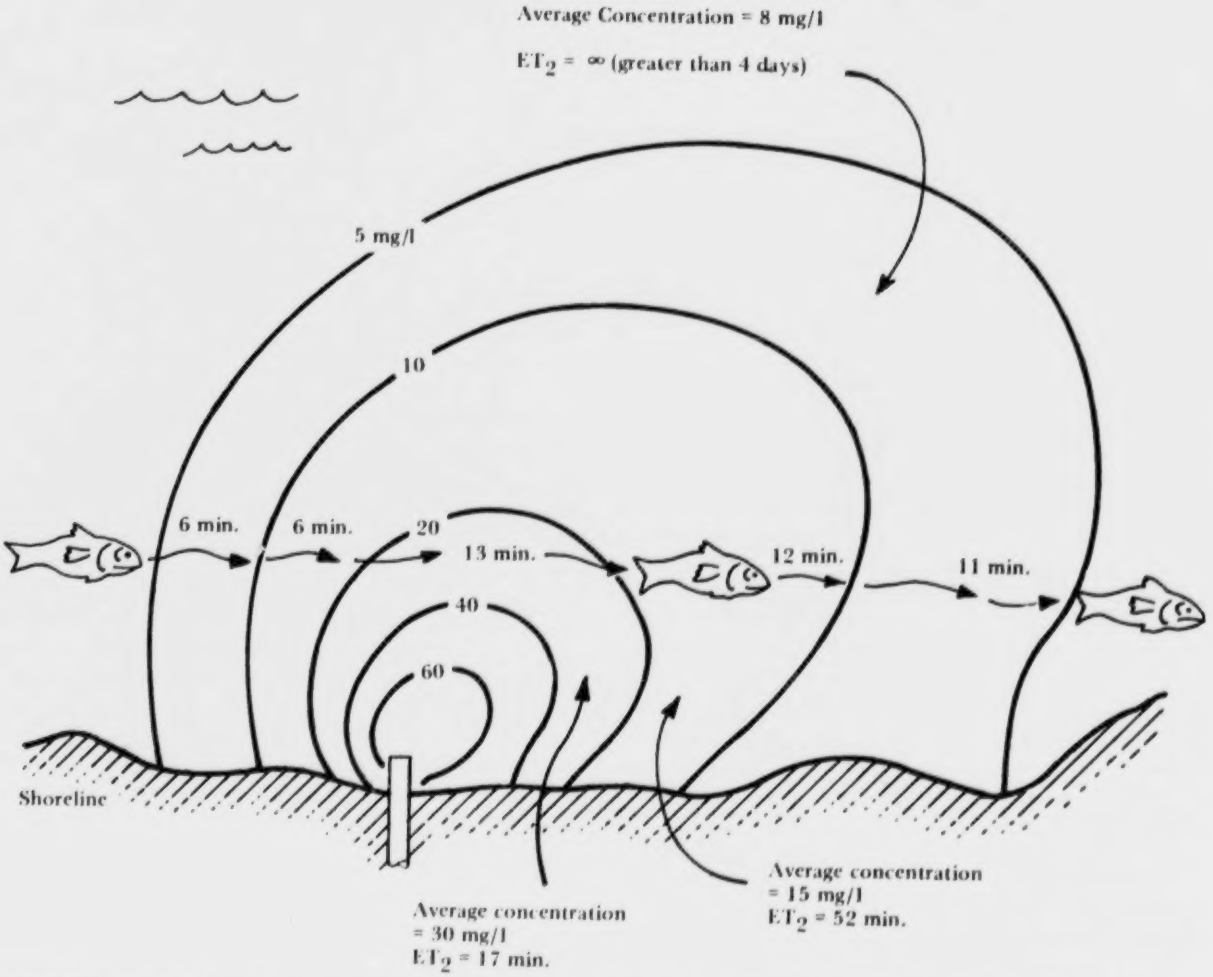

FIGURE II-A-1—Predicted Concentrations of ABS in an Effluent Plume, and Times of Passage of Migrating Fish. Hypothetical.

Comments

a. Use of the ET(X). A probability distribution is involved in mortality, and it is therefore impossible to give any valid estimate of an exposure time which would cause zero per cent mortality. The probability of mortality merely becomes increasingly smaller as the exposure time becomes less. Therefore it is necessary to choose some arbitrary percentage mortality as equivalent to negligible effect. Two per cent was chosen as a useful level in the example above since it is a low number yet still high enough that the extrapolation of the probit line to that value has reasonable validity. Other mortality levels can be selected to fit given situations.

When mortality is the response measured rather than a more conservative one, a safety factor can be utilized by requiring the sum of the integrated time-exposure effects to equal less than unity.

b. Toxicity Curves. For other toxicants, the curves may be greatly different from those shown in Figure II-A-3, e.g., complex reflex or rectangular hyperbolas. Further discussion of toxicity curves, and illustration of curves of various shapes is given by Warren (1971,[34] pp. 199–203) and Sprague (1969).[28]

It is possible to calculate equations for the toxicity curves, or portions of them, as was done for temperature-mortality data (pp. 151 ff.). However, the equations for many toxicants are cumbersome because of logarithms or other transformation. Since the equations are merely the result of empirically fitting the observed experimental curves, it is easier and about equally effective to read values of interest directly from a graph such as Figure II-A-3.

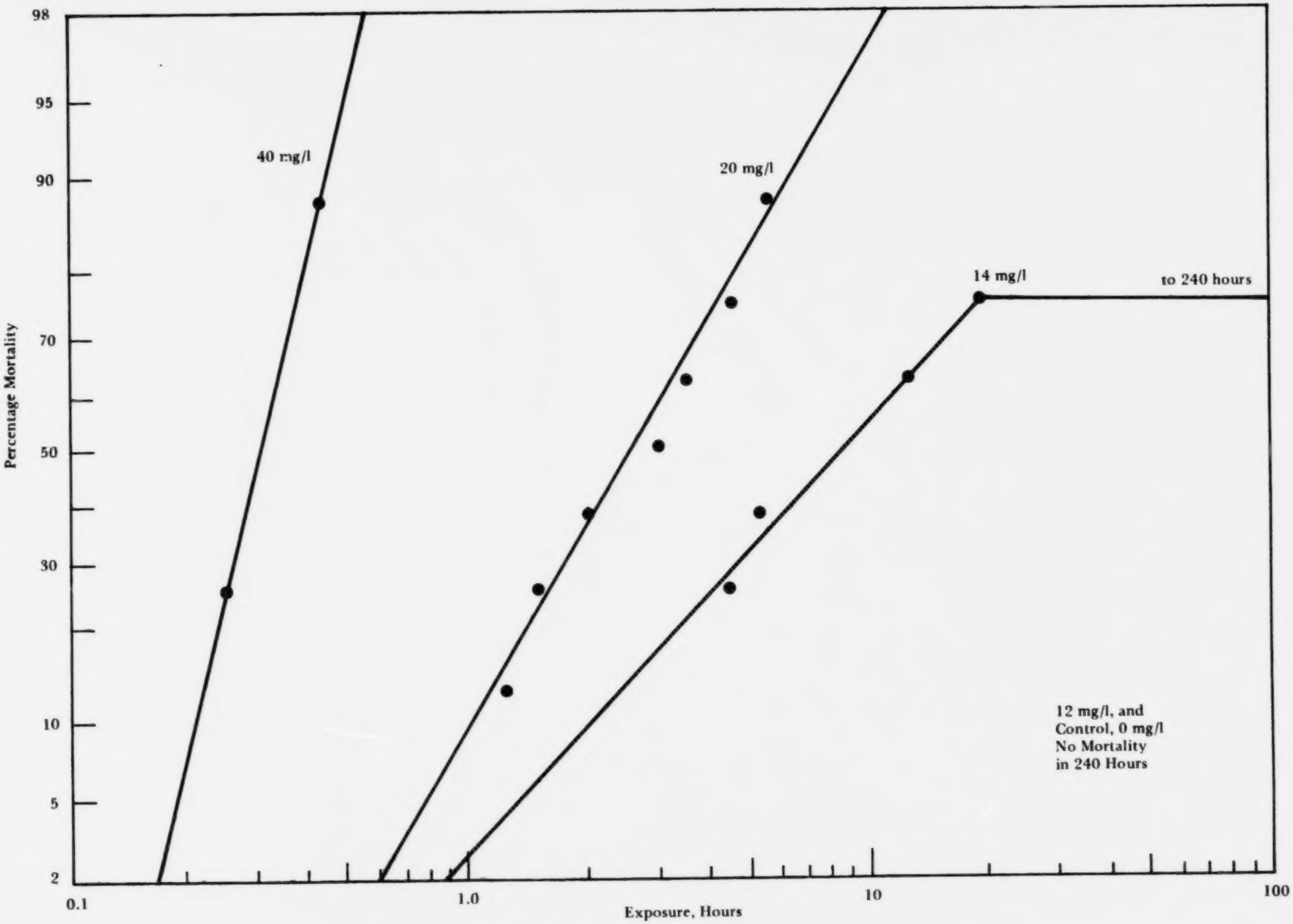

FIGURE II-A-2—Mortality of Rainbow Trout Exposed to Concentrations of ABS.

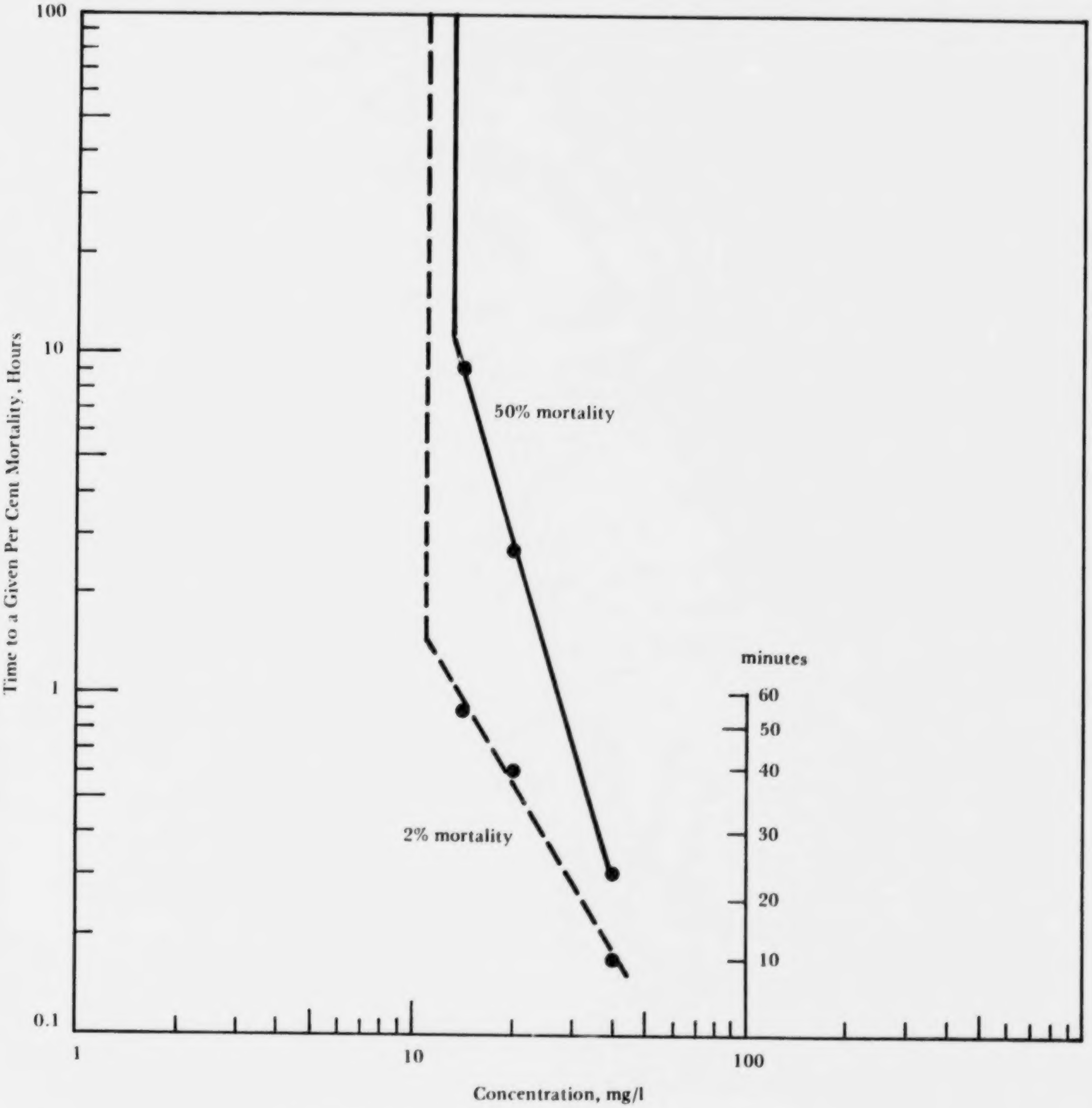

FIGURE II–A–3—Toxicity Curves for ABS to Rainbow Trout.

The times to 50 per cent mortality and times to 2 per cent mortality have been read from the lines fitted in Figure II–A–2.

c. Threshold Effective Time. Organisms may survive for 30 minutes, an hour, or sometimes several hours, even in extremely high concentrations of the pollutant (see caveat under d).

d. Lethal Threshold Concentration. Survival for an indefinitely long period may be possible at the lethal threshold concentration which may be close to concentrations which are quickly lethal. Organisms which exhibit an abrupt lethal threshold or a long threshold effective time may be especially vulnerable to sublethal effects and careful investigation of this possibility should be made.

e. Need for Experimental Determination of ET(X). Although it would be convenient to have some rule of thumb for estimating the ET(X) from the ET50, as is done by the "2° rule" for short-term exposure to high temperature (see Section III, pp. 161–162), there does not seem to be any such simple generalization which can be applied to toxicants in general. The relatively few examples which can be found in the literature indicate variable relationships. A series of comparisons between toxicity curves for 5 per cent and 50 per cent mortality are given by Herbert (1961,[14] 1965[15]) and Herbert and Shurben (1964).[16] The ratios between LC5 and LC50 for the same exposure times are as follows: fluoride 0.4; a demulsifier 0.55; ammonium chloride 0.55 (high concentrations) and 0.8 (low concentrations); washing powders 0.75, and a corrosion inhibitor 0.88. Even for the same pollutant the ratio is different for different concentrations when the time-concentration relation is curved, as it is for many substances. A difference is also found when the toxicity curves are not parallel, as for ABS in Figure II-A-3. The LC2/LC50 ratio for ABS varies from 0.46 to 0.72 at high concentrations and short times, and increases to 0.87 for the 96-hour exposures.

Because of this variability, no simple rule of thumb can be proposed for estimating, from the 50 per cent values, the concentrations which will produce negligible mortality or the exposure times for negligible mortality. It is necessary to determine this empirically by the steps used in constructing Figure II-A-2.

APPENDIX II-B

COMMUNITY STRUCTURE AND DIVERSITY INDICES

Evaluation Systems for Protection

There are two basic approaches in evaluating the effects of pollution on aquatic life: the first by a taxonomic grouping of organisms; the second by identifying the community of aquatic organisms.

First, the saprobian system of Kolkwitz and Marsson (1908,[19] 1909[20]), modified and used by Richardson (1928),[63] Gaufin (1956),[44] Hynes (1962)[48] and Beck (1954,[38] 1955[39]), depended upon a taxonomic grouping of organisms related to their habitat in clean water, polluted water, or both. This approach requires a precise identification of organisms. It is based on the fact that different organisms have different ranges of tolerance to the same stress. Patrick (1951)[59] and Wurtz (1955)[67] used a system of histograms to report the results of stream surveys based on the differences in tolerance of various groups of aquatic organisms to pollution. Beck (1955)[39] developed a biotic index as a method of evaluating the effects of pollution on bottom fauna organisms. The biotic index is calculated by multiplying the number of intolerant species by two and then adding the number of facultative organisms. Beck designated a biotic index value greater than 10 to indicate clean water and a value less than 10 to indicate polluted water. Other techniques based on the tolerance of aquatic organisms to pollution have been reported by Gaufin (1958)[45] and Beak (1965).[37]

The breakdown of an assemblage of organisms into pollution-tolerant, -intolerant, and -facultative categories is somewhat subjective, because tolerance for the same organisms may vary under a different set of environmental conditions. Needham (1938)[58] observed that environmental conditions other than pollution may influence the distribution of organisms. Pollution-tolerant organisms are also found in clean water areas (Gaufin and Tarzwell, 1952).[46] Therefore, the concept of the use of taxonomic groupings of organisms to evaluate water quality biologically has certain difficulties and is not commonly accepted today.

The second approach is to use the community structure of associations or populations of aquatic organisms to evaluate pollution. Hairston (1959)[47] defined community structure in terms of frequency of species per unit area, spatial distribution of individuals, and numerical abundance of species. Gaufin (1956)[44] found that the community structure of benthic invertebrates provided a more reliable criterion of organic enrichment than presence of a specific species.

Diversity indices that permit the summarization of large amounts of information about the numbers and kinds of organisms have begun to replace the long descriptive lists common to early pollution survey work. These diversity indices result in a numerical expression that can be used to make comparisons between communities of organisms. Some of these have been developed to express the relationships of numbers of species in various communities and overlap of species between communities.

The Jaccard Index is one of the commonest used to express species overlap. Other indices such as the Shannon-Weiner information theory (Shannon and Weaver 1963)[64] have been used to express the evenness of distribution of individuals in species composing a community. The diversity index increases as evenness increases (Margalef 1958,[52] Hairston 1959,[47] MacArthur and MacArthur 1961,[55] and MacArthur 1964[53]). Various methods have been developed for comparing the diversity of communities and for determining the relationship of the actual diversity to the maximum or minimum diversity that might occur within a given number of species. Methods have been thoroughly discussed by Lloyd and Ghelardi (1964),[51] Patten (1962),[60] MacArthur (1965),[54] Pielou (1966,[61] 1969[62]), McIntosh (1967)[57], Mathis (1965)[56], Wilhm (1965),[65] and Wilhm and Dorris (1968)[66] as to what indices are appropriate for what kinds of samples. An index for diversity of community structure also has been developed by Cairns, Jr. et al. (1968)[40] and Cairns, Jr. and Dickson (1971)[41] based on a modification of the sign test and theory runs of Dixon and Massey (1951).[42]

Diversity indices derived from information theory were first used by Margalef (1958)[52] to analyze natural communities. This technique equates diversity with information. Maximum diversity, and thus maximum information, exists in a community of organisms when each individual belongs to a different species. Minimum diversity (or high redundancy) exists when all individuals belong to the same species. Thus, mathematical expressions can be used for diversity and redundancy that describe community structure.

As pointed out by Wilhm and Dorris (1968),[66] natural biotic communities typically are characterized by the presence of a few species with many individuals and many species with a few individuals. An unfavorable limiting factor such as pollution results in detectable changes in community structure. As it relates to information theory, more information (diversity) is contained in a natural community than in a polluted community. A polluted system is simplified, and those species that survive encounter less competition and therefore may increase in numbers. Redundancy in this case is high, because the probability that an individual belongs to a species previously recognized is increased, and the amount of information per individual is reduced.

The relative value of using indices or models to interpret data depends upon the information sought. To see the relative distribution of population sizes among species, a model is often more illuminating than an index. To determine information for a number of different kinds of communities, diversity indices are more appropriate. Many indices overemphasize the dominance of one or a few species and thus it is often difficult to determine, as in the use of the Shannon-Weiner information theory, the difference between a community composed of one or two dominants and a few rare species, or one composed of one or two dominants and one or two rare species. Under such conditions, an index such as that discussed by Fisher, Corbet and Williams (1943)[43] is more appropriate. To use the Shannon-Weiner index, much more information about the community is obtained if a diversity index is plotted.

This section is the basis for the criteria on change of diversity given in the sections on Suspended Solids and Hardness, Temperature, and Dissolved Oxygen.

APPENDIX II-C

THERMAL TABLES—Time-temperature relationships and lethal threshold temperatures for resistance of aquatic organisms (principally fish) to extreme temperatures (from Coutant, in press[75] 1972). Column headings, where not self-explanatory, are identified in footnotes. LD50 data obtained for single times only were included only when they amplified temperature-time information.

Species	Stage age	Length	Weight	Sex	Location	Reference	Extreme	Acclimation		log time = a + b (temp.)				Data limits (°C)		LD50	Lethal threshold[d] (°C)
								Temp[a]	Time	a	b	N[b]	r[c]	upper	lower		
Abudefduf saxatilis (Sargent major)	Adult				Northern Gulf of California	Heath, W. G. (1967)[89]	Upper	32		42.9005	−0.0934	3	−0.9945	37.0	36.0		
Adinia xenica (diamond Killifish)	Adult				Jefferson Co., Texas	Strawn and Dunn (1967)[99]	Upper	35	(0 ‰)[e]	21.9337	−0.4866	6	−0.9930	43.0	40.5		
								35	(5 ‰)[e]	27.7919	−0.6159	6	−0.9841	43.5	41.0		
								35	(10 ‰)[e]	26.8121	−0.5899	6	−0.9829	43.5	41.0		
								35	(20 ‰)[e]	28.3930	−0.6290	6	−0.9734	43.5	41.0		
Atherinops affinis (topsmelt)	Juvenile	6.0-6.2 cm			LaJolla, Calif.	Doudoroff (1945[79])	Upper	18.0								30.5(24)	
								20		42.2531	−1.2215	9	−0.9836	33.5	31.5		31.0
							Lower	14.5								7.6(24)	
								18.0								8.8(24)	
								20		−0.4667	0.3926	7	0.9765	11.0	5.0		10.5
								25.5								13.5(24)	
Brevoortia tyrannus (Atlantic menhaden)	Larval	17–34 mm		Mixed	Beaufort Harbor, North Carolina (36°N)	Lewis (1965)[91]	Lower	7.0		0.9611[f]	0.2564	9	0.9607	4.0			5.0
							″	10.0		0.7572	0.2526	12	0.9452	5.0	−1.0		6.0
							″	12.5		0.6602	0.2786	12	0.9852	5.5			>7.0
							″	15.0		0.5675	0.2321	14	0.9306	7.0			>8.0
							″	20.0		0.2620	0.1817	3	0.9612	4.0			
Brevoortia tyrannus (Atlantic menhaden)	Young-of-the-year				Beaufort; N.C.	Lewis and Hettler (1968)[92]	Upper	21	(5 ‰)	57.9980	−0.1643	2		35.0	34.0		
								27	(5 ‰)	85.1837	−2.3521	2		35.0	34.5		
							Lower	16	(26-30 ‰)					7.0	3.0		6.5
								18	(10 ‰)					7.0	3.0		6.5
Brevoortia tyrannus (Atlantic menhaden)	Yearling				Beaufort, N.C.	Lewis and Hettler (1968)[92]	Upper	21	(5 ‰)	35.7158	−1.0488	3	−0.9174	34	33		
								22-23	(4-6 ‰)	21.8083	−0.6342	10	−0.9216	35	31		32.5
Crassius aura (goldfish)	Juvenile		2g ave.	Mixed	Commercial dealer (Toronto)	Fry, Brett, & Clawson (1942)[81] (and Fry, Hart, & Walker, 1946)[83]	Upper	1-2								28 (14)	
								10								31 (14)	
								17								34 (14)	
								24								36 (14)	
								32		20.0213	−0.4523	2		41.0	39.0	39.2(14)	
								38		21.9234	−0.4773	2		43.0	41.0	41.0(14)	41.0
							Lower	19								1.0(14)	
								24								5.0(14)	
								38								15.5(14)	
Catostomus commersonni (white sucker)	Adult (1-2 yr)		10-19.9 (mode)	Mixed	Don River, Thornhill, Ontario	Hart (1947[87])	Upper	5		33.6957	−1.1797	2		27.5	27.0		26.3
								10		19.9890	−0.6410	3	−0.6857	29	28		27.7
								15		31.9007	−1.0034	2		30	29.5		29.3
								20		27.0023	−0.8068	4	−0.9606	31.5	30		29.3
								25		22.2209	−0.6277	7	−0.9888	32.5	29.5		29.3
							Lower	20									2.5
								25									6.0

[a] It is assumed in this table that the acclimation temperature reported is a true acclimation in the context of Brett (1952).[74]

[b] Number of median resistance times used for calculating regression equation.

[c] Correlation coefficient (perfect fit of all data points to the regression line = 1.0).

[d] = Incipient lethal temperature of Fry, et al., (1946).[83]

[e] Salinity.

[f] Log time in hours to 50% mortality. Includes 2-3 hr. required for test bath to reach the test temperature.

THERMAL TABLES—Continued

Species	Stage/age	Length	Weight	Sex	Location	Reference	Extreme	Acclimation		log time=a+b (temp.)				Data limits (°C)		LD50	Lethal threshold[d] (°C)
								Temp[a]	Time	a	b	N[b]	r[c]	upper	lower		
Coregonus artedii (cisco)	Juvenile	……	……	Mixed	Pickerel Lake,[e] Washtenaw Co., Mich.	Edsall and Colby, 1970[102]	Upper	2	8 wks	16.5135	−0.6682	4	−0.9789	23.0	19.0	……	19.7
								5	4 wks	10.2789	−0.3645	3	−0.9264	24.0	20.0	……	21.7
								10	>2 wks	12.4993	−0.4098	6	−0.9734	28.0	24.0	……	24.2
								20	2 wks	17.2967	−0.5333	8	−0.9487	30.0	26.0	……	26.2
								25	3 wks	15.1204	−0.4493	7	−0.9764	30.0	25.5	……	25.7(u)
							Lower	2	8 wks	……	……	……	……	1.5	0.3	……	<0.3
								5	4 wks	……	……	……	……	1.0	0.5	……	<0.5
								10	>2 wks	2.7355	0.3381	5	0.9021	3.0	0.5	……	3.0
								20	2 wks	2.5090	0.2685	6	0.9637	4.5	0.5	……	4.7
								25	3 wks	1.7154	0.1652	9	0.9175	9.5	0.5	……	9.7
Coregonus hoyi (bloater)	Juvenile (age 1)	60.0 mm 5.0. 5.8	……	Mixed	Lake Michigan at[f] Kenosha, Wisc.	Edsall, Rottiers & Brown, 1970[80]	Upper	5	11 da[g]	15.8243	−0.5831	5	−0.9095	26.0	22.0	……	22.2
								10	5 da	9.0700	−0.2896	6	−0.9516	30.0	23.0	……	23.6
								15	5 da	17.1908	−0.5707	4	−0.9960	28.0	24.5	……	24.6
								20	5 da	28.6312	−0.9458	4	−0.9692	29.0	25.5	……	26.2
								25	5 da	21.3511	−0.6594	5	−0.9958	30.0	26.5	……	26.7
Cyprinodon variegatus (sheepshead minnow)	Adult	……	……	……	Jefferson County, Texas	Strawn and Dunn (1967[99])	Upper	35	(0 ‰)	27.9021	−0.6217	6	−0.9783	43.0	40.5	……	40.5
								35	(5 ‰)	35.3415	−0.7858	6	−0.9787	43.5	41.0		
								35	(10 ‰)	30.0910	−0.8629	6	−0.9950	43.5	41.5		
								35	(20 ‰)	30.0394	−0.6594	4	−0.9982	43.5	41.5		
Cyprinodon variegatus variegatus (sheepshead minnow)	Adult	……	……	……	Galveston Island, Galveston, Texas	Simmons (1971)[97]	Upper	30	700 hrs.[h] (from 21.3 C)	35.0420	−0.8025	2	……	41.4	40.8	……	……
Dorosoma cepedianum (gizzard shad)	Underyearling	……	……	……	Put-in-Bay, Ohio	Hart (1952)[66]	Upper	25	field & 3–4 da	47.1163	−1.3010	3	−0.9975	35.5	34.5	……	34.0
								30	"	38.0858	−0.9694	4	−0.9921	38.0	36.5	……	36.0
								35	"	31.5434	−0.7710	5	−9.9642	39.0	37.0	……	36.5(u)
							Lower	25	……	……	……	……	……	……	……	……	10.8
								30	……	……	……	……	……	……	……	……	14.5
								35	……	……	……	……	……	……	……	……	20.0
Dorosoma cepedianum (gizzard shad)	Underyearling	……	……	……	Knoxville, Tenn.	Hart (1952)[66]	Upper	25	……	32.1348	−0.8698	2	……	35.5	35.0	……	34.5
								30	……	41.1030	−0.0547	4	−0.9981	38.0	36.5	……	36.0
								35	……	33.2846	−0.8176	6	−0.9896	39	36.5	……	36.5
Esox lucius (Northern Pike)	Juvenile	Minimum 5.0 cm	……	……	Maple, Ontario, Canada	Scott (1964)[96]	Upper	25.0	……	17.3066	−0.4523	5	−0.9990	34.5	32.5	……	32.25
								27.5	……	17.4439	−0.4490	5	−0.9985	35.0	33.0	……	32.75
								30.0	……	17.0961	−0.4319	5	−0.9971	35.5	33.5	……	33.25(u)
Esox masquinongy (Muskellunge)	Juvenile	Minimum 5.0 cm	……	……	Deerlake Hatchery Ontario, Canada	Scott (1964)[96]	Upper	25.0	……	18.88[illegible]	−0.5035	5	−0.9742	34.5	32.5	……	32.25
								27.5	……	20.081[illegible]	−0.5283	5	−0.9911	35.0	33.0	……	32.75
								30.0	……	18.9506	−0.4851	5	−0.9972	35.5	33.5	……	33.25 (u)
Esox hybrid (luciusx masquinongy)	Juvenile	5.0 cm minimum	……	……	Maple, Ontario, Canada	Scott (1964)[96]	Upper	25.0	……	18.6533	−0.4926	4	−0.9941	34.5	33.0	……	32.5
								27.5	……	20.7834	−0.5460	5	−0.9995	35.0	33.0	……	32.75
								30.0	……	19.6126	−0.5032	5	−0.9951	35.5	33.5	……	33.25 (u)
Fundulus chrysotus (golden topminnow	Adult	……	……	……	Jefferson County, Texas	Strawn & Dunn (1967)[99]	Upper	35	(0 ‰)—	23.7284	−0.5219	9	−0.9968	43.0	39.0	……	38.5
								35	(5 ‰)—	21.2575	−0.4901	7	−0.9969	43.5	40.0	……	……
								35	(20 ‰)—	21.8635	−0.4759	8	−0.9905	43.5	40.0	……	……
Fundulus diaphanus (banded killifish)	Adult	……	……	……	Halifax Co. and Annapolis Co., Nova Scotia	Garside and Jordan (1968)[84]	Upper	15	(0 ‰)[i]	……	……	……	……	……	……	……	27.5
								15	(14 ‰)	……	……	……	……	……	……	……	33.5
								15	(32 ‰)	……	……	……	……	……	……	……	27.5
Fundulus grandis (gulf killifish)	Adult	……	……	……	Jefferson County, Texas	Strawn & Dunn (1967)[99]	Upper	35	(0 ‰)	22.9809	−0.5179	8	−0.9782	42.0	38.5	……	……
								35	(5 ‰)	27.6447	−0.6220	7	−0.9967	42.5	39.5	……	……
								35	(10 ‰)	24.9072	−0.5535	9	−0.9926	43.0	39.0	……	……
								35	(20 ‰)	23.4251	−0.5199	8	−0.9970	43.0	39.5	……	……
Fundulus heteroclitus (mummichog)	Adult	……	……	……	Halifax Co. and Annapolis Co., Nova Scotia	Garside and Jordan (1968)[84]	Upper	15	(0 ‰)[j]	……	……	……	……	……	……	……	28.0
								15	(14 ‰)	……	……	……	……	……	……	……	34.0
								15	(32 ‰)	……	……	……	……	……	……	……	31.5

[a] It is assumed in this table that the acclimation temperature reported is a true acclimation in the context of Brett (1952).[74]

[b] Number of median resistance times used for calculating regression equation.

[c] Correlation coefficient (perfect fit of all data points to the regression line=1.0).

[d] = Incipient lethal temperature of Fry, et al., (1946).[83]

[e] Experimental fish were hatched from eggs obtained from adults from this location.

[f] Experimental fish were reared from eggs taken from adults from this location.

[g] These times after holding at 8 C for >1 mo.

[h] Acclimated and tested at 10 ‰ salinity.

[i] Tested in three salinities.

[j] Tested at 3 levels of salinity.

THERMAL TABLES—Continued

Species	Stage/age	Length	Weight	Sex	Location	Reference	Extreme	Acclimation		log time=a+b (temp.)				Data limits (°C)		LD50	Lethal threshold[d] (°C)
								Temp[a]	Time	a	b	N[b]	r[c]	upper	lower		
Fundulus parvipinnis (California killifish) (tested in seawater except as noted)	Adult	6-7 cm	....	Mixed	Mission Bay, Calif. (seawater)	Doudoroff (1945)[79]	Upper	14	....	23.3781	−0.6439	4	−0.9845	34.0	32.0	....	32.3
								20	....	50.6021	−1.3457	11	−0.9236	37.0	34.0	....	34.4
								28	....	24.5427	−0.5801	7	−0.9960	40.0	38.0	....	36.5
							Lower	14	....	2.1908	1.0751	3	0.9449	1.6	0.4	....	1.2
								20	....	2.7381	0.2169	6	0.9469	7.0	2.0	....	5.6
								20	....	2.5635	0.3481	4	0.8291	4.0	2.0	....	3.6
								20	(into 45% sea water 1 day before testing)	2.6552	0.4014	8	0.7348	4.0	2.0	....	3.8
Fundulus pulvereus (bayou killifish)	Adult	....	....	....	Jefferson County, Texas	Strawn and Dunn (1967)[99]	Upper	35	(0 ‰)	28.1418	−0.6304	8	−0.9741	43.0	39.0	....	38.5
								35	(5 ‰)	29.3774	−0.6514	7	−0.9931	43.5	40.0	....	....
								35	(10 ‰)	25.0890	−0.5477	5	−0.9956	43.5	41.5	....	....
								35	(20 ‰)	30.4702	−0.6745	8	−0.9849	43.5	40.0	....	....
Fundulus similis (longnose killifish)	Adult	....	....	....	Jefferson County, Texas	Strawn and Dunn (1967)[99]	Upper	35	(0 ‰)[e]	22.9485	−0.5113	6	−0.9892	43.0	40.5	....	....
								35	(5 ‰)	25.6105	−0.5690	6	−0.9984	43.5	41.0	....	....
								35	(10 ‰)	26.4675	−0.5863	6	−0.9925	43.5	41.0	....	....
								35	(20 ‰)	26.5012	−0.5879	6	−0.9953	43.0	40.5	....	....
Gambusia affinis affinis (mosquitofish)	Adult	....	....	Mixed	Knoxville, Tenn.	Hart (1952)[88]	Upper	25	....	39.0004	−0.9771	2	.....	39	38	....	37.0
								30	....	30.1523	−0.7143	6	−0.9938	40	37.5	....	37.0
								35	....	23.8110	−0.5408	6	−0.9978	41.5	39	....	37.0(u)
Gambusia affinis (mosquitofish) (freshwater)	Adult	....	....	....	Jefferson Co., Texas	Strawn & Dunn (1967)[99]	Upper	35	(0 ‰)[e]	22.4434	−0.5108	5	−0.9600	42.0	40.0	....	....
								35	(5 ‰)	23.1338	−0.5214	5	−0.9825	42.5	40.5	....	....
								35	(10 ‰)	23.4977	−0.5304	8	−0.9852	42.5	40.0	....	....
								35	(20 ‰)	22.1994	−0.5001	6	−0.9881	42.5	40.0	....	....
Gambusia affinis (mosquitofish) (saltwater)	Adult	....	....	....	Jefferson Co., Texas	Strawn and Dunn (1967)[99]	Upper	35	(0 ‰)[e]	17.6144	−0.3909	5	−0.9822	42.5	40.5	....	....
								35	(5 ‰)	18.9339	−0.4182	5	−0.9990	42.5	40.5	....	....
								35	(10 ‰)	23.0784	−0.5165	7	−0.9982	42.5	39.5	....	....
								35	(20 ‰)	22.8863	−0.5124	6	−0.9957	42.5	40.0	....	....
Gambusia affinis holbrooki (mosquitofish)	Adult	....	....	Mixed	Welaka, Florida	Hart (1952)[88]	Upper	15	....	32.4692	−0.8507	3	−0.9813	37	36	....	35.5
								20	....	38.3139	−0.9673	3	−0.9843	38.5	37.5	....	37.0
								30	....	31.4312	−0.7477	5	−0.9995	40	38	....	37.0
								35	....	28.1212	−0.6564	5	−0.9909	40	38.5	....	37.0(u)
							Lower	15	....	....	....	.....	....	....	.....	....	1.5
								20	....	....	....	.....	....	....	.....	....	5.5
								35	....	....	....	.....	....	....	.....	....	14.5
Garmannia chiquita (goby)	Adult	....	....	....	Northern Gulf of California Coast	Heath (1967)[89]	Upper	32	....	21.7179	−0.5166	3	−0.9905	37.0	36.0	....	....
Gasterosteus aculeatus (three-spine stickleback)	Adult	37 mm ave.	0.50 g ave.	Mixed	Columbia River near Prescott, Oregon	Blahm and Parente (1970)[101] unpublished data	Upper	19	....	19.3491	−9.5940	3	−0.9998	32	26	....	25.8
Girella nigricans (opaleye)	Juvenile	7.1-8.0 cm	....	Mixed	LaJolla, California (33°N)	Doudoroff (1942)[78]	Upper	12	....	21.1277	−0.6339	6	−0.9338	31.0	27.0	....	28.7
								20	....	19.2041	−0.5080	7	−0.9930	35.0	31.0	....	31.4
								28	....	24.7273	−0.6740	4	−0.9822	33.0	31.0	....	31.4
							Lower	12	....	1.4851	0.4886	8	0.9550	5.0	1.0	....	5.5
								20	....	−1.3878	0.6248	6	0.9895	8.0	5.0	....	8.5
								28	....	−0.1238	0.2614	6	0.9720	13.0	6.0	....	13.5
Ictalurus (Amicurus) nebulosus (brown bullhead)	....	....	....	....	Florida to Ontario (4 locations) combined	Hart (1952)[88]	Upper	5	....	14.6802	−0.4539	4	−0.9782	29.5	28.0	....	27.8
								10	....	16.4227	−0.4842	10	−0.9526	31.5	29.5	....	29.0
								15	....	28.3281	−0.8239	3	−0.9861	33.0	32.5	....	31.0
								20	....	23.9586	−0.6473	11	−0.9712	35.0	32.5	....	32.5
								25	....	22.4970	−0.5732	12	−0.9794	37.0	34.0	....	33.8
								30	....	24.2203	−0.5917	19	−0.9938	38.5	35.5	....	34.8
								34	....	19.3194	−0.4500	5	−0.9912	37.5	36.0	....	34.8
							Lower	20	....	....	....	.....	....	....	.....	....	0.5
								25	....	....	....	.....	....	....	.....	....	4.0
								30	....	....	....	.....	....	....	.....	....	6.8
Ictalurus punctatus (channel catfish)	Juvenile (44-57 da old)	....	....	Mixed	Centerton, Ark. (hatchery)	Allen & Strawn (1968)[72]	Upper	26	....	34.7119	−0.8816	13	−0.9793	39.0	36.6	....	36.6
								30	....	32.1736	−0.7811	17	−0.9510	40.6	37.4	....	37.8
								34	....	26.4204	−0.5149	20	−0.9638	42.0	38.0	....	38.0

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[b] Number of median resistance times used for calculating regression equation.

[c] Correlation coefficient (perfect fit of all data points to the regression line=1.0).

[d] = Incipient lethal temperature of Fry, et al., (1946).[83]

[e] Salinity.

THERMAL TABLES—Continued

Species	Stage/age	Length	Weight	Sex	Location	Reference	Extreme	Acclimation		log time=a+b (temp.)				Data limits (°C)		LD50	Lethal threshold[d] (°C)
								Temp[a]	Time	a	b	N[b]	r[c]	upper	lower		
Ictalurus punctatus (channel catfish)	Juvenile (11.5 mo)	....	....	....	Joe Hogan State Fish Hatchery, Lonoke, Arkansas	Allen & Strawn (1968)[73]	Upper	25	....	34.5554	0.8854	5	−0.9746	37.5	35.5	....	35.5
								30	....	17.7125	−0.4058	4	−0.9934	40.0	37.5	....	37.0
								35	....	28.3031	−0.6554	4	−0.9906	41.0	38.0	....	38
Ictalurus punctatus (I. lacustris) (channel catfish)	Adult	....	....	Mixed	Welaka, Fla. and Put-in-Bay, Ohio	Hart (1952)[88]	Upper	15	....	34.7829	−1.0637	3	−0.9999	31.5	30.5	....	30.4
								20	....	39.4967	−1.1234	4	−0.9980	34.0	33.0	....	32.8
								25	....	46.2155	−1.2899	5	−0.9925	35.0	34.0	....	33.5
							Lower	15	....	....	....	.....	....	....	.....	....	0.0
								20	....	....	....	.....	....	....	.....	....	0.0
								25	....	....	....	.....	....	....	.....	....	0.0
Lepomis macrochirus purpurescens (bluegill)	Adult	....	....	Mixed	Welaka, Florida	Hart (1952)[88]	Upper	15	....	25.2708	−0.7348	5	−0.9946	33.0	31.0	....	30.5
								20	....	28.0663	−0.7826	6	−0.9978	34.5	32.5	....	32.0
								25	....	23.8733	−0.6320	10	−0.9750	36.0	33.0	....	33.0
								30	....	25.7732	−0.6581	5	−0.9965	38	34.5	....	33.8
							Lower	15	....	....	....	.....	....	....	.....	....	2.5
								20	....	....	....	.....	....	....	.....	....	5.0
								25	....	....	....	.....	....	....	.....	....	7.5
								30	....	....	....	.....	....	....	.....	....	11.0
Lepomis macrochirus (bluegill)	Adult	....	....	Mixed	Lake Mendota, Wisconsin	Hart (1952)[88]	Upper	20–23	....	38.6247	−1.0581	4	−0.8892	35.5	34.0	....	....
								30	....	30.1609	−0.7657	4	−0.9401	38.0	36.0	....	....
Lepomis megalotis (longear sunfish)	Juvenile	>12 mm	....	Mixed	Middle Fork, White River, Arkansas	Neill, Strawn & Dunn (1966)[95]	Upper	25	....	35.4953	−0.9331	14	−0.9827	36.9	35.4	....	35.6
								30	....	20.5981	−0.4978	22	−0.9625	39.0	36.5	....	36.8
								35	....	30.7245	−9.7257	43	−0.9664	41.5	37.3	....	37 5
Lepomis symmetricus (bantam sunfish)	Adult	....	....	....	Jefferson Co., Texas	Strawn & Dunn (1967)[99]	Upper	35	(0 °/oo)[e]	20.7487	−0.4686	7	−0.9747	42.0	39.0	....	....
								35	(5 °/oo)	23.5649	−0.5354	6	−0.9975	42.0	39.0	....	....
								35	(20 °/oo)	10.4421	−0.2243	5	−0.9873	41.5	39.5	....	....
Lucania parva (rainwater killifish)	Adult	....	....	....	Jefferson Co., Texas	Strawn and Dunn (1967)[99]	Upper	35	(0 °/oo)[e]	21.2616	−0.4762	9	−0.9844	42.5	38.5	....	....
								35	(5 °/oo)	24.3076	−0.5460	8	−0.9846	42.5	39.0	....	....
								35	(10 °/oo)	24.3118	−0.5467	8	−0.9904	42.5	39.0	....	....
								35	(20 °/oo)	21.1302	−0.4697	7	−0.9840	42.5	39.5	....	....
Menidia menidia (common silverside)	....	8.3–9.2 cm (average for test groups)	4.3–5.2 gm (average for test groups)	Mixed	New Jersey (40°N)	Hoff & Westman (1966)[90]	Upper	7	....	19.8801	−0.7351	5	−0.9398	24.0	20	....	22.0
								14	....	18.7499	−0.6001	6	−0.9616	27.0	23.0	....	25.0
								21	....	65.7356	−2.0387	6	−0.9626	32.0	28.0	....	30.4
								28	....	37.6032	−1.0582	5	−0.8872	34.0	30	....	32.5
							Lower	7	....	−9.8144	8.9079	5	0.8274	2	1	....	1.5
								14	....	−1.2884	2.5597	6	0.8594	5	1	....	2.0
								21	....	−1.4801	1.1484	6	0.9531	7	2	....	4.3
								28	....	−8.2366	1.3586	5	0.9830	15	7	....	8.7
Micropterus salmoides floridanus (largemouth bass)	9–11 mo. age	....	....	....	Welaka, Florida	Hart (1952)[88]	Upper	20	....	35.5107	−1.0112	5	−0.9787	34	32	....	32
								25	....	19.9918	−0.5123	8	−0.9972	36.5	33	....	33
								30	....	17.5645	−0.4200	8	−0.9920	38	34.5	....	33.7(u)
							Lower	20	....	....	....	.....	....	....	.....	....	5.2
								25	....	....	....	.....	....	....	.....	....	7.0
								30	....	....	....	.....	....	....	.....	....	10.5
Micropterus salmoides (largemouth bass)	....	....	....	....	Put-in-Bay, Ohio	Hart (1952)[88]	Upper	20	....	50.8091	−1.4638	2	....	34	33	....	32.5
								25	....	26.3169	−0.6846	3	−0.9973	36.5	35	....	34.5
								30	....	29.0213	−0.7150	4	−0.9959	38.5	37	....	36.4(u)
							Lower	20	....	....	....	.....	....	....	.....	....	5.5
								30	....	....	....	.....	....	....	.....	....	11.8
Micropterus salmoides (largemouth bass)	Under yearling	....	....	....	Knoxville, Tenn.	Hart (1952)[88]	Upper	30	....	36.0620	−0.9055	4	−0.9788	38.5	37	....	36.4
								35	....	23.9185	−0.5632	6	−0.9958	40	37.5	....	36.4(u)
Micropterus salmoides (largemouth bass)	....	....	....	....	Lake Mendota, Wisconsin	Hart (1952)[88]	Upper	22	....	34.3649	−0.9789	4	−0.9789	33.8	32.0	....	31.5
								30	....	35.2777	−0.9084	4	−0.9845	37.5	35.5	....	....
Mysis relicta (Opposum shrimp)	Adult	....	....	Mixed	Trout Lake, Cook County, Minnesota	Smith (1970)[98]	Upper	7.5C	>1 wk	6.1302	−0.1470	3	0.9245	26	16	....	16

[a] It is assumed in this table that the acclimation temperature reported is a true acclimation in the context of Brett (1952).[74]

[b] Number of median resistance times used for calculating regression equation.

[c] Correlation coefficient (perfect fit of all data points to the regression line=1.0).

[d] = Incipient lethal temperature of Fry, et al., (1946).[83]

[e] Salinity.

THERMAL TABLES—Continued

Species	Stage/age	Length	Weight	Sex	Location	Reference	Extreme	Acclimation		log time=a+b (temp.)				Data limits (°C)		LD50	Lethal threshold[d] (°C)
								Temp[a]	Time	a	b	N[b]	r[c]	upper	lower		
Neomysis awatschensis (opossum shrimp)	Adult	>7 mm	...	Mixed	Sacramento-San Joaquin delta, California	Hair (1971)[86]	Upper	10.3[e]	...	...	...	...	...	...	...	73 (48)	...
								11.0	...	...	...	...	...	...	...	72.5(48)	...
								15.1	...	...	...	...	...	...	...	73.8(48)	...
								18.3	...	...	...	...	...	...	...	76.1(48)	...
								19.0	...	...	...	...	...	...	...	74.0(48)	...
								19.0	...	8.4094	−0.2150	2	...	...	...	...	24.2–25.4[f]
								21.7	...	...	...	...	...	...	...	77.0(48)	...
								22.0	...	...	...	...	...	...	...	77.5(48)	...
								22.4	...	...	...	...	...	...	...	76.0(48)	...
Notemigonus crysoleucas (golden shiner)	Adult	...	...	...	Composite[g] of 1. Welaka, Fla. 2. Put-in-Bay, Ohio 3. Algonquin Park, Ontario	Hart (1952)[88]	Upper	10	...	42.7095	−1.3507	3	−0.9998	30.5	29.5	...	29.5
								15	...	30.2861	−0.8933	4	−0.9844	32.5	31.0	...	30.5
								20	...	31.0275	−0.8722	15	−0.9809	34.5	32.0	...	32.0
								25	...	34.2505	−0.9226	9	−0.9665	36.0	34	...	33.5
								30	...	26.3829	−0.6615	10	−0.9940	37.5	35	...	34.5
							Lower	15	...	...	...	...	...	...	...	...	1.5
								20	...	...	...	...	...	...	...	...	4.0
								25	...	...	...	...	...	...	...	...	7.0
								30	...	...	...	...	...	...	...	...	11.2
Notropis atherinoides (emerald shiner)	Juvenile (<1yr)	...	0–1.9 g.mode	Mixed	Chippewa Creek, Welland, Ontario	Hart (1947)[87]	Upper	5	...	20.9532	−0.7959	3	−0.9519	24.5	23.5	...	23.3
								10	...	36.5023	−1.2736	2	...	27.5	27.0	...	26.7
								15	...	47.4849	−1.5441	3	−0.9803	30.5	29.5	...	28.9
								20	...	33.4714	−0.9858	3	−0.9805	32.5	31.5	...	30.7
								25	...	25.7096	−0.7337	6	−0.9753	34.0	31.5	...	30.7
							Lower	15	...	...	...	...	...	...	...	...	1.6
								20	...	...	...	...	...	...	...	...	5.2
								25	...	...	...	...	...	...	...	...	8.0
Notropis cornutus (common shiner)	Adult	...	...	...	Toronto, Ontario	Hart (1952)[88]	Upper	10	...	...	...	1	...	29.0	29.0	...	29.0
								15	...	45.4331	−1.3979	2	...	31.5	31.0	...	30.5
								20	...	34.5324	−1.0116	4	−0.9560	33.0	31.5	...	31.0
								25(winter)	...	24.9620	−0.6878	5	−0.9915	34.0	32.0	...	31.0
								25	...	28.5059	−0.7741	8	−0.9973	35.5	32.0	...	31.0
								30	...	28.1261	−0.7316	6	−0.9946	36.5	34.0	...	31.0(u)
Notropis cornutus (common shiner)	Adult (mostly 2 yr)	...	4.0–5.9 g (mode)	Mixed	Don River, Thornhill, Ontario	Hart (1947)[87]	Upper	5	...	...	...	...	...	...	...	...	26.7
								10	...	40.7738	−1.3522	3	−0.9729	30.0	29.0	...	28.6
								15	...	45.0972	−1.3874	3	−0.9990	32.0	31.0	...	30.3
								20	...	34.5324	−1.0116	4	−0.9560	33.0	31.5	...	31.0
								25	...	24.9620	−0.6878	5	−0.9915	34.0	32.0	...	31.0
							Lower	20	...	...	...	...	...	...	...	...	3.7
								25	...	...	...	...	...	...	...	...	7.8
Notropis cornutus (common shiner)	Adult	...	...	...	Knoxville, Tenn.	Hart (1952)[88]	Upper	25	...	25.5152	−0.6794	6	−0.9938	35.5	33.0	...	33.0
								30	...	24.9660	−0.6297	10	−0.9978	38.0	34.5	...	33.5(u)
Oncorhynchus gorbuscha (pink salmon)	Juvenile freshwater fry (3.8 mo.)	3.81±0.29 cm	0.30±0.15g	Mixed	Dungeness, Wash. (hatchery)	Brett (1952)[74]	Upper	5	...	11.1827	−0.4215	4	−0.9573	24.0	22.0	...	21.3±0.3
								10	...	11.9021	−0.3865	8	−0.9840	26.5	23.0	...	22.5±0.3
								15	...	12.8937	−0.4074	8	−0.9884	27.0	23.5	...	23.1±0.3
								20	...	16.2444	−0.4074	7	−0.9681	27.5	24.0	...	23.9±0.6
								24	...	14.7111	−0.4459	6	−0.9090	27.5	24.5	...	23.9
Oncorhynchus keta (chum salmon)	Juvenile freshwater fry (4.9 mo.)	5.44±0.89 cm	1.62±1.03g	Mixed	Nile Creek, B.C. (hatchery)	Brett (1952)[74]	Upper	5	...	14.3829	−0.5320	4	−0.9839	24.0	22.0	...	21.8
								10	...	14.1773	−0.4766	9	−0.8665	26.5	22.5	...	22.6
								15	...	15.8911	−0.5252	8	−0.9070	27.0	23.0	...	23.1±0.4
								20	...	16.1894	−0.5168	9	−0.9750	27.5	23.5	...	23.7
								23	...	15.3825	−0.4721	4	−0.9652	27.0	24.0	...	23.8±0.4
							Lower	5	...	...	...	...	...	...	...	...	...
								10	...	...	...	...	...	1	...	...	0.5
								15	...	...	...	...	...	5	...	...	4.7
								20	...	...	...	...	...	7	...	...	6.5
								23	...	...	...	...	...	8	...	...	7.3
Oncorhynchus keta (chum salmon)	Juvenile	...	...	...	Big Creek Hatchery, Hoodsport, Wash.[h]	Blahm and Parente (1970)[101] unpublished data	Upper	9	10%[i]	16.9245	−0.5995	6	−0.9927	29	17	...	22.0
									50%	15.9272	−0.5575	4	−0.9972	29	17	...	23.2
									90%	16.8763	−0.5881	4	−0.9995	29	17	...	23.6

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[a] It is assumed in this table that the acclimation temperature reported is a true acclimation in the context of Brett (1952).[74]

[b] Number of median resistance times used for calculating regression equation.

[c] Correlation coefficient (perfect fit of all data points to the regression line=1.0).

[d] =Incipient lethal temperature of Fry, et al., (1946).[83]

[e] All temperatures estimated from a graph.

[f] For maximum of 48 hr exposure. The lower temperature is uncorrected for heavy mortality of control animals at "acclimation" temperatures above about 21.6.

[g] The author concluded that there were no geographic differences. The Welaka, Florida subspecies was N.c. bosii, the others N.c. auratus, based on morphology.

[h] Tested in Columbia River Water at Prescott, Oregon.

[i] Mortality Value.

THERMAL TABLES—Continued

Species	Stage/age	Length	Weight	Sex	Location	Reference	Extreme	Acclimation		log time=a+b (temp.)				Data limits (°C)		LD50	Lethal threshold[d] (°C)
								Temp[a]	Time	a	b	N[b]	r[c]	upper	lower		
Oncorhynchus Kisutch (coho salmon)	Juvenile fresh-water fry (5.2 mo.)	4.78±0.6 cm	1.37±0.62g	Mixed	Nile Creek, B.C. (hatchery)	Brett (1952)[74]	Upper	5	....	21.3050	−0.7970	2	....	24.0	23.0	....	22.9±0.3
								10	....	19.5721	−0.6820	4	−0.9847	26.0	24.5	....	23.7
								15	....	20.4066	−0.6858	6	−0.9681	27.0	24.5	....	24.3±0.3
								20	....	20.4022	−0.6713	4	−0.9985	27.5	25.5	....	25.0±0.2
								23	....	18.9736	−0.6013	5	−0.9956	27.5	25.0	....	25.0±0.2
							Lower	5	....	....	....	.....	....	.....	.....	....	0.2
								10	....	....	....	.....	....	1	.....	....	1.7
								15	....	....	....	.....	....	3	.....	....	3.5
								20	....	....	....	.....	....	5	.....	....	4.5
								23	....	....	....	.....	....	7	1.0	....	6.4
Oncorhynchus kisutch (coho salmon)	Juvenile	....	....	....	Kalama Falls, Wash. (hatchery)[e]	Blahm & McConnell (1970)[100] unpublished data	Upper	10	(10%)[f]	15.4616	−0.5522	6	−0.8533	29	1.7	....	23.2
									(50%)	18.4136	−0.6410	6	−0.9705	29	17.0	....	23.5
									(90%)	15.9026	−0.5423	4	−0.9720	29	17.0	....	23.7
								14[g]	(10%)	8.5307	−0.2969	10	−0.9063	29	14.0	....	14.0
									(50%)	8.5195	−0.2433	10	−0.8483	29	0.14	....	17.0
									(90%)	....	....	.....	....	.....	.....	....	22.0
Oncorhynchus kisutch (coho salmon)	Adult	a 570 mm ave.	a 2500 g ave.	Mixed	Columbia River at Priest Rapids Dam	Coutant (1970)[76]	Upper	17[h]	....	5.9068	−0.1630	5	−0.9767	30	26	....	?
Oncorhynchus nerka (sockeye salmon)	Juvenile fresh-water fry (4.7 mo)	4.49±0.84 cm	0.87±0.45g	Mixed	Issaquah, Wash. (hatchery)	Brett (1952)[74]	Upper	5	....	17.7887	−0.6623	4	−0.9383	24.0	22.5	....	22.2±0.3
								10	....	14.7319	−0.4988	8	−0.9833	26.5	23.5	....	23.4±0.3
								15	....	15.8799	−0.5210	7	−0.9126	27.5	24.5	....	24.4±0.3
								20	....	19.3821	−0.6378	5	−0.9602	27.5	24.5	....	24.8±0.3
								23	....	20.0020	−0.6496	4	−0.9981	26.5	24.5	....	24.8±0.3
							Lower	5	....	....	....	.....	....	0	0	....	0
								10	....	....	....	.....	....	4	0	....	3.1
								15	....	....	....	.....	....	5	0	....	4.1
								20	....	....	....	.....	....	5	0	....	4.7
								23	....	....	....	.....	....	7	1.0	....	6.7
Oncorhynchus nerka (sockeye salmon)	Juvenile (under yearling)	67 mm ave.		Mixed	National Fish Hatchery[i] Leavenworth, Wash.	McConnell & Blahm (1970)[103] unpublished data	Upper	10	10%[j]	18.4771	−0.6458	6	−0.9671	29	17	....	21.5
									50%	18.5833	−0.6437	6	−0.9750	29	17	....	22.5
									90%	20.6289	−0.7166	6	−0.9553	29	17	....	23.0
								20	10%	17.5227	−0.5861	6	−0.9739	29	21	....	23.5
									50%	16.7328	−0.5473	6	−0.9552	29	21	....	23.5
									90%	15.7823	−0.5061	6	−0.9539	29	21	....	23.5
Oncorhynchus nerka (sockeye salmon)	Juvenile (yearline)	100–105 mm are for test groups		Mixed	National Fish Hatchery Leavenworth, Wash.[i]	McConnell & Blahm (1970)[103] unpublished data	Upper	10 1°C per day rise to accl. temp.	(10%)[j]	6.4771	−0.2118	4	−0.9887	32	14		
									(50%)	9.0438	−0.2922	4	−0.9392	32	14	....	23.5
									(90%)	9.0628	−0.2859	4	−0.9534	32	14	....	....
								12″	(10%)	13.2412	−0.4475	4	−0.9955	29	17	....	....
									(50%)	18.1322	−0.6178	4	−0.9598	29	17	....	23.5
									(90%)	17.5427	−0.5900	4	−0.9533	29	17	....	....
								15.5″	(10%)	12.1763	−0.4004	5	−0.9443	32	17	....	....
									(50%)	13.6666	−0.4432	5	−0.9720	32	17	....	22.5
									(90%)	12.7165	−0.4057	4	−0.9748	32	17	....	....
								17″	(10%)	17.4210	−0.6114	5	−0.9549	29	20	....	....
									(50%)	17.2432	−0.5885	4	−0.9450	29	20	....	23.5
									(90%)	17.2393	−0.5769	4	−0.9364	29	20	....	....
Oncorhynchus tshawytscha (Chinook salmon)	Juvenile fresh-water fry (3.6 mo.)	4.44±0.40 cm	1.03±0.27g	Mixed	Dungeness, Wash. (hatchery)	Brett (1952)[74]	Upper	5	....	9.3155	−0.3107	6	−0.9847	25.0	22.5	....	21.5
								10	....	16.4595	−0.5575	5	−0.9996	26.5	24.5	....	24.3±0.1
								15	....	16.4454	−0.5364	4	−0.9906	27.0	25.5	....	25.0±0.1
								20	....	22.9065	−0.7611	7	−0.9850	27.5	25.0	....	25.1±0.1
								24	....	18.9940	−0.5992	9	−0.9923	27.5	25.0	....	25.1±0.1
							Lower	10	....	....	....	.....	....	1.0	0	....	0.8
								15	....	....	....	.....	....	3.0	0.5	....	2.5
								20	....	....	....	.....	....	5.0	0.5	....	4.5
								23	....	....	....	.....	....	8.0	1.0	....	7.4

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[a] It is assumed in this table that the acclimation temperature reported is a true acclimation in the context of Brett (1952).[74]

[b] Number of median resistance times used for calculating regression equation.

[c] Correlation coefficient (perfect fit of all data points to the regression line = 1.0).

[d] = Incipient lethal temperature of Fry, et al., (1946).[83]

[e] 10 C—acclimated fish came directly from the hatchery.

[f] Data were presented allowing calculation of 10% and 90% mortality.

[g] 14 C—acclimated fish were collected from the Columbia River 4-6 wks following release from the hatchery (and may have included a few fish from other upstream sources). River water was supersaturated with Nitrogen, and 14-C fish showed signs of gas-bubble disease during tests.

[h] River temp. during fall migration.

[i] Tested in Columbia River water at Prescott, Oregon.

[j] Per cent mortalities.

THERMAL TABLES—Continued

Species	Stage/age	Length	Weight	Sex	Location	Reference	Extreme	Acclimation		log time=a+b (temp.)				Data limits (°C)		LD50	Lethal threshold[d] (°C)
								Temp[a]	Time	a	b	N[b]	r[c]	upper	lower		
Oncorhynchus tshawytscha (chinook salmon)	Juvenile	30-124 mm averages for various test groups	....	Mixed	Columbia River at Prescott, Oregon	Snyder & Blahm (1970)[105] unpublished data	Upper	10[e]	....	16.8109	−0.5787	3	−0.9998	29	25	....	24.5
									(10%[f])	18.9770	−0.6621	5	−0.9918	29	23	....	22.9
									(90%)	17.0278	−0.5845	3	−0.9997	29	25	....	24.5
								10[g]	....	15.7101	−0.5403	8	−0.9255	29	20	....	23.5
									(10%)	15.1583	−0.5312	8[h]	−0.9439	29	20	....	20.5
									(90%)	15.2525	−0.5130	8	−0.9360	29	20	....	23.5
								12	....	18.2574	−0.6149	5[h]	−0.9821	29	23	....	20.5
								13	....	12.4058	−0.3974	6	−0.9608	32	17	....	20.0
									(10%)	10.1410	−0.3218	7	−0.9496	32	17	....	19.5
									(90%)	12.7368	−0.4040	6	−0.9753	32	17	....	23.0
								18[g]	....	13.3175	−0.4240	11	−0.9550	30	20	....	20.5
									(10%)	11.5122	−0.3745	12	−0.9413	30	20	....	20.0
									(90%)	14.2456	−0.4434	10	−0.9620	30	20	....	23.5
Oncorhynchus tshawytscha (Chinook salmon spring run)	Juvenile	84 mm ave.	6.3g ave.	Mixed	Little White Salmon, River Hatchery, Cook, Washington	Blahm & McConnell (1970)[100] unpublished data	Upper	11	2-3-wks								
									10%[i]	13.3696	−0.4691	4	−0.9504	29	17	....	23.0
									50%	14.6268	−0.5066	4	−0.9843	29	17	....	23.5
									90%	19.2211	−0.6679	4	−0.9295	29	17	....	23.8
								20	1C/day rise from 10C								
									10%	22.6664	−0.7797	4	−0.9747	29	21	....	23.8
									50%	21.3981	−0.7253	3	−0.9579	29	21	....	24.7
									90%	20.9294	−0.7024	3	−0.9463	29	21	....	24.8
Oncorhynchus tshawytscha (chinook salmon)	Juvenile	40 mm. ave.	....	Mixed	Eggs from Seattle, Wash. raised from yolk-sac stage in Columbia River water at Prescott, Oregon	Snyder & Blahm (1970)[105] unpublished data	Upper	4	....	13.5019	−0.4874	4	−0.9845	29	8	....	20
									(10%)[j]	8.9126	−0.3198	6	−0.9618	29	8	....	13.5
									(90%)[j]	10.6491	−0.3771	6	−0.9997	29	8	....	?
Oncorhynchus tshawytscha (chinook salmon fall run)	Juvenile	90.6 mm ave.	7.8 g ave.	Mixed	Little White Salmon Riverhatchery, Cook, Washington	Blahm & McConnell (1970)[100] unpublished data	Upper	11	2-3 wks								
									10%[k]	18.6889	−0.6569	5	−0.9618	29	17	....	23.5
									50%	20.5471	−0.7147	4	−0.9283	29	17	....	24.2
									90%	20.8960	−0.7231	4	−0.9249	29	17	....	24.5
							Upper	20	1C/day rise from 10C								
									10%	21.6756	−0.7438	4	−0.9550	29	21	....	24.5
									50%	22.2124	−0.7526	4	−0.9738	29	21	....	24.5
									90%	20.5162	−0.6860	3	−0.9475	29	21	....	24.5
Oncorhynchus tshawytscha (Chinook salmon)	"Jacks" 1-2 yrs old	2500 mm ave.	2000 g. ave.	Males	Columbia River at Grand Rapids Dam	Coutant (1970)[76]	Upper	17[l]	....	13.2502	−0.4121	4	−0.8206	30	26	....	?
								19[l]	....	9.4683	−0.2504	4	−0.9952	26	22	....	22
Perca flavescens (yellow perch)	Juvenile	48 mm ave.	1.2 g ave.	Mixed	Columbia River near Prescott, Ore.	Blahm and Parente (1970)[101] unpublished data	Upper	19	field plus 4 da.	15.3601	−0.4126	2	....	38	32	....	?
Perca flavescens (yellow perch)	Adult (4 yr mode)	....	8.0-9.9 g mode	Mixed	Black Creek, Lake Simcoe, Ontario	Hart (1947)[87]	Upper	5	....	7.0495	−0.2214	9	−0.9904	26.5	22.0	....	21.3
								11	....	17.6536	−0.6021	2	....	26.5	26.0	....	25.0
								15	....	12.4149	−0.3641	5	−0.9994	30.5	28.5	....	27.7
								25	....	21.2718	−0.5909	6	−0.9698	33.0	30.0	....	29.7
							Lower	25	....	....	....	....	....	....	....	....	3.7
Petromyzon marinus (sea lamprey, land-locked)	Prolarvae	....	....	....	Great Lakes	McCauley (1963)[94]	Upper	15 and 20[m]	....	17.5642	−0.4680	18	−0.9683	34	29	....	28.5

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[a] It is assumed in this table that the acclimation temperature reported is a true acclimation in the context of Brett (1952).[74]

[b] Number of median resistance times used for calculating regression equation.

[c] Correlation coefficient (perfect fit of all data points to the regression line=1.0).

[d] = Incipient lethal temperature of Fry, et al., (1946).[83]

[e] Fish tested shortly after capture by beach seine.

[f] Data were also available for calculation of 10% and 90% mortality of June test groups.

[g] These were likely syhergistic effects of high N2 supersaturation in these tests.

[h] Excluding apparent long-term secondary mortality.

[i] Data were available for 10% and 90% mortality as well as 50%.

[j] Data also available on 10% and 90% mortality.

[k] Data available for 10% and 90% mortality as well as 50%.

[l] River temperatures during fall migrations two different years.

[m] No difference was shown so data are lumped.

THERMAL TABLES—Continued

Species	Stage/age	Length	Weight	Sex	Location	Reference	Extreme	Acclimation		log time = a+b (temp.)				Data limits (°C)		LD50	Lethal threshold[d] (°C)
								Temp[a]	Time	a	b	N[b]	r[c]	upper	lower		
Pimephales (Hyborhynchus) notatus (bluntnose minnow)	Adult (mostly 1 yr)	...	mostly 0–2 g	Mixed	Etobicoke Cr., Ontario	Hart (1947)[87]	Upper	5	...	24.6417	−0.8602	2	...	27.0	26.5	...	28.0
								10	...	55.8357	−1.8588	2	...	29.5	25.0	...	28.3
								15	...	28.0377	−0.6337	3	−0.9974	32.0	31.0	...	30.6
								20	...	34.3240	−0.9682	4	−0.9329	34.0	32.5	...	31.7
								25	...	50.8212	−1.4181	3	−0.9480	35.0	34.0	...	33.3
							Lower	15	...	...	...	...	...	...	...	...	10
								20	...	...	...	...	...	...	...	...	4.2
								25	...	...	...	...	...	...	...	...	7.5
Pimephales promelas (fathead minnow)	Adult (1 yr)	...	2.0–3.9 g mode	Mixed	Don River, Thornhill, Ontario	Hart (1947)[87]	Upper	10	...	60.7782	−2.0000	2	...	30.0	29.5	...	28.2
								20	...	6.9970	−0.1560	4	−0.7448	33.0	28.5	...	31.7
								30	...	41.3096	−1.1317	5	−0.9670	36.0	34.0	...	33.2
							Lower	20	...	...	...	...	...	...	...	...	1.5
								30	...	...	...	...	...	...	...	...	10.5
Poecilia latipinna (Sailfin molly)	Adult				Jefferson Co., Texas	Strawn and Dunn (1967)[99]	Upper	35	(0 ‰)[e]	27.4296	−0.6279	6	−0.9902	42.5	38.5	...	...
								35	(5 ‰)	25.6936	−0.5753	6	−0.9835	42.5	39.0	...	...
								35	(10 ‰)	28.8808	−0.6535	7	−0.9949	42.0	39.0	...	...
								35	(20 ‰)	27.1988	−0.6146	3	−0.9791	42.5	39.5	...	...
Pontoporeia affinis	Adult			Mixed	Lake Superior near Two Harbors, Minn.	Smith (1971)[116] unpublished data	Upper	6	...	9.1790	−0.5017	2	...	12	10.8	...	10.5
								9	...	...	...	...	...	...	...	10.4 (30 ds)	...
Pseudopleuronectes americanus (winter flounder)		6.0–7.1 cm (averages for test groups)	3.4–4.2 g (averages for test groups)	Mixed	New Jersey (40°N)	Hoff & Westman (1966)[90]	Upper	7	...	28.2986	−1.1405	4	−0.9852	24.0	20.0	...	22.0
								14	...	24.3020	−0.8762	6	−0.9507	26.0	23.0	...	23.7
								21	...	40.0231	−1.6915	5	−0.9237	29.0	26.0	...	27.0
								28	...	60.8070	−1.9610	4	−0.9181	30.0	29.0	...	29.1
							Lower	7	...	...	...	...	...	1.0	1.0	...	1.0
								14	...	...	...	...	...	2.0	1.0	...	1.0
								21	...	2.4924	0.8165	3	0.7816	6.0	1.0	...	14
								28	...	2.2145	0.2344	3	0.9970	7.0	4.0	...	6.0
Rhinichthys atratulus (blacknose dace)	Adult				Knoxville, Tenn.	Hart (1952)[88]	Upper	20	...	21.2115	−0.5958	7	−0.9935	33	30	...	29.3
								25	...	19.6451	−0.5224	10	−0.9979	35	30.5	...	29.3
								28	...	21.3360	−0.5651	7	−0.9946	35.5	32.5	...	29.3
Rhinichthys atratulus (blacknose dace)	Adult (?)				Toronto, Ontario	Hart (1952)[88]	Upper	5	...	...	...	...	...	27	27	27(1 hr)	...
								15	...	19.8158	−0.5771	4	−0.9632	31.5	30.0	...	29.3
								20	...	24.5740	−0.7061	7	−0.9926	33	30.0	...	29.3
								25	...	20.1840	−0.5389	8	−0.9968	35	32.0	...	29.3
Rhinichthys atratulus (Blacknose dace)	Adult		2.0–3.9 (mode)	Mixed	Don River, Thornhill, Ontario	Hart (1947)[87]	Upper	5	...	77.1877	−2.7959	2	...	27.5	27.0	...	26.5
								10	...	49.1469	−1.6021	3	−0.8521	30.5	29.5	...	28.8
								15	...	19.6975	−0.5734	4	−0.9571	31.5	30.0	...	29.6
								20	...	26.5952	−0.7719	8	−0.9897	33.5	29.5	...	29.3
								25	...	23.5765	−0.6629	9	−0.9937	34.0	30.0	...	29.3
							Lower	20	...	...	...	...	...	...	...	...	2.2
								25	...	...	...	...	...	...	...	...	5.0
Salmo gairdnerii (Rainbow trout)	Juvenile	4.5±0.4 cm		Mixed	Britain	Alabaster & Welcomme (1962)[70]	Upper	18[f]	...	18.4654	−0.5801	5	−0.9787	29.6	26.3	...	26.5
								18[g]	...	13.6531	−0.4264	5	−0.9742	29.1	26.3	...	26.5
Salmo gairdnerii (rainbow trout)	Yearling	...	...	...	East end of Lake Superior	Craigie, D.E. (1963)[77]	Upper	Raised in soft water 20 (tested in soft water)		14.6405	−0.4470	3	−0.9787	29	27	...	...
								20 (tested in hard water)		15.0392	−0.4561	3	−0.9917	29	27	...	...
								Raised in hard water 20 (tested in soft water)		15.1473	−0.4683	3	−0.9781	29	27	...	...
								20 (tested in hard water)		12.8718	−0.3837	3	−0.9841	29	27	...	...
Salmo gairdnerii (rainbow trout)	Juvenile	9.4±6.0 cm and 15.5±1.8 cm	...	Mixed	London, England (Hatchery)	Alabaster & Downing (1966)[69]	Upper	15	...	15.6500	−0.500	2[h]	...	...	...	...	...
								20	...	19.6250	−0.6250	2	...	...	...	...	...

[a] It is assumed in this table that the acclimation temperature reported is a true acclimation in the context of Brett (1952).[74]

[b] Number of median resistance times used for calculating regression equation.

[c] Correlation coefficient (perfect fit of all data points to the regression line = 1.0).

[d] = Incipient lethal temperature of Fry, et al., (1946).[83]

[e] Salinity.

[f] Dissolved oxygen Conc. 7.4 mg/l.

[g] Dissolved oxygen Conc. 3.8 mg/l.

[h] See note (under Salmo salar) about Alabaster 1967.[66]

THERMAL TABLES—Continued

Species	Stage/age	Length	Weight	Sex	Location	Reference	Extreme	Acclimation		log time = a+b (temp.)				Data limits (°C)		LD50	Lethal threshold[d] (°C)
								Temp[a]	Time	a	b	N[b]	r[c]	upper	lower		
Salmo gairdnerii (anadromous) (Steelhead trout)	Adult	2650 mm ave.	4000 g ave.	Mixed	Columbia River at Priest Rapids Dam	Coutant (1970)[76]	Upper	19[e]	....	10.9677	−0.3329	7	−0.9910	29	21	....	21
Salmo salar (Atlantic salmon)	Smolts (1–2 yrs)	About 16 cm ave.	....	Mixed	River Axe, Devon, England	Alabaster (1967)[69]	Upper	9.2 (field)		43.6667	−1.6667	2[f]	....	(f)	(f)	....	....
								9.3"		23.7273	−0.9091	2	....	....	....	....	....
								10.9"		126.5000	−5.000						
								Tested in 30% seawater									
								9.2 (field)		44.6667	−1.6667	2	....	....	....	....	....
								Tested in 100% seawater									
								9.2 (field)		14.7368	−0.5263	2	....	....		....	....
								Acclimated 7 hr in seawater; tested in seawater									
								9.2 (field)		36.9999	−1.4286	2	....	....	....	....	....
Salmo salar (Atlantic salmon)	Newly hatched larvae	....	....	Mixed	Cullercoats North Shields, England (hatchery)	Bishai (1960)[73]	Upper	6 (brought up to test temp. in 6 hours)		13.59	−0.4287	6	−0.9678	28.0	20.0	....	22.0
Salmo salar (Atlantic salmon)	30 da after hatching	....	....	Mixed	Cullercoats, North Shields, England (hatchery)	Bishai (1960)[73]	Upper	5	....	8.9631	−0.2877	4	−0.9791	25.0	22	....	22.2
								10	....	15.7200	−0.5396	3	−0.9689	26.0	22	....	23.3
								20	....	11.5471	−0.3406	3	−0.9143	26.0	22	....	23.5
Salmo salar (Atlantic salmon)	Parr (1 yr)	10 cm ave.	....	Mixed	River Axe, Devon, England	Alabaster (1967)[69]	Upper	9.3 (field)		33.3750	−1.2500	2[g]	....	....	....	....	....
								10.9 (field)		28.0000	−1.0000	2	....	....	....	....	....
Salmo salar (Atlantic salmon)	Smolts (1–2 yrs)	11.7±1.5 cm	....	Mixed	River North Esk, Scotland	Alabaster (1967)[69]	Upper	11.7	....	25.9091	−0.9091	2[g]	....	....	....	....	....
Salmo salar (Atlantic salmon)	Smolts (1–2 yrs)	14.6±1.3 cm	....	Mixed	River Severn Gloucester, England	Alabaster (1967)[69]	Upper	16.7	....	14.5905	−0.4545	2[g]	....	....	....	....	....
Salmo trutta (brown trout)	Newly hatched fry	....	....	Mixed	Cullercoats, North Shields, England (hatchery)	Bishai (1960)[73]	Upper	6 (raised to test temp. over 6 hr period)		12.7756	−0.4010	6	−0.9747	28.0	20.0	....	22.0
Salmo trutta (Brown trout, searun)	30 da after hatching	....	....	Mixed	Cullercoats, North Shields, England (hatchery)	Bishai (1960)[73]	Upper	5	....	15.2944	−0.5299	4	−0.8783	25.0	22.0	....	22.2
								10	....	23.5131	−0.8406	3	−0.9702	26.0	22.0	....	23.4
								20	....	14.6978	−0.4665	3	−0.9797	26.0	22.0	....	23.5
Salmo trutta (brown trout, searun)	Juvenile	10.1±0.8 cm 7.4±4.5 cm	....	Mixed	London, England (hatchery)	Alabaster & Downing (1966)[69]	Upper	6	....	36.1429	−1.4286	2[g]	....	....	....	....	....
								15	....	21.5714	−0.7143	2	....	....	....	....	....
								20	....	17.6667	−0.5556	2	....	....	....	....	....
Salmo trutta (brown trout, searun)	Smolts (2 yr.)	About 21 cm ave.	....	Mixed	River Axe, Devon, England	Alabaster (1967)[69]	Uppe	9.3 (field)		18.4667	−0.6667	2[g]	....	....	....	....	....
								10.9"		33.0000	−1.2500	2	....	....	....	....	....
Salvelinus fontinalis (Brook trout)	Juvenile	....	....	....	Pleasant Mount Hatchery, Wayne Co., Penna. and Chatsworth Hatchery, Ontario[h]	McCauley (1958)[85]	Upper	10	....	17.5260	−0.6033	6	−0.9254	25.5	24.5	....	....
								20	....	20.2457	−0.6671	7	−0.9723	27.0	25.0	....	....

[a] It is assumed in this table that the acclimation temperature reported is a true acclimation in the context of Brett (1952).[74]

[b] Number of median resistance times used for calculating regression equation.

[c] Correlation coefficient (perfect fit of all data points to the regression line = 1.0).

[d] = Incipient lethal temperature of Fry, et al., (1946).[83]

[e] River temp. during fall migration

[f] Alabaster fitted by eye, a straight line to median death times plotted on semilog paper (log time), then reported only the 100 and 1000 min intercepts. These intercepts are the basis for the equation presented here.

[g] See note for Alabaster 1967.[69]

[h] Results did not differ so data were combined.

THERMAL TABLES—Continued

Species	Stage/age	Length	Weight	Sex	Location	Reference	Extreme	Acclimation		log time = a+b (temp.)				Data limits (°C)		LD50	Lethal threshold[d] (°C)
								Temp[a]	Time	a	b	N[b]	r[c]	upper	lower		
Salvelinus fontinalis (brook trout)	Yearling		$\bar{X}$ = 7.88 g range 2–25 g	Mixed	Codrington, Ont. (hatchery	Fry, Hart & Walker (1946)[83]	Upper	3		13.4325	−0.4556	3	−0.9997	26.0	23.5		23.5
								11		14.6256	−0.4728	6		28.0	25.0		24.6
								15		15.1846	−0.4833	9		28.5	25.5		25.0
								20		15.0331	−0.4661	7		29.0	25.5		25.3
								22		17.1967	−0.5367	6		29.0	26.5		25.5
								24		17.8467	−0.5567	10		30.0	25.5		25.5
								25		17.8467	−0.5567	3		29.0	26.0		25.5
Salvelinus fontinalis (namaycush hybrid)	Juvenile				Ontario, Canada	Fry and Gibson (1953)[82]	Upper	10		13.2634	−0.4381	6	−0.9852	26.5	24.0		23.5–24.0
								15		16.9596	−0.5540	8	−0.9652	28.0	24.5		?
								20		19.4449	−0.6342	9	−0.9744	28.0	24.5		24.0–24.5
Salvelinus namaycush (Lake trout)	1–2 yr. old		27.7 gm ave. (1 yr) 82.8 gm ave. (2 yr)	Mixed	Hatcheries in Ontario	Gibson and Fry (1954)[85]	Upper	8	1 wk	14.4820	−0.5142	4	−0.9936	26	23		22.7
								15	"	14.5123	−0.4866	5	−0.9989	27	24		23.5
								20	"	17.3684	−0.5818	5	−0.9951	27	24		23.5
Scardinius erythrophthalamus (rudd)	Adult	10 cm		Mixed	Britain (field)	Alabaster & Downing (1966)[69]	Upper	20		26.9999	−0.7692	2[e]					
Semotilus atromaculatus (Creek chub)	Adult		2.0–3.9 gm mode	Mixed	Don River, Thornhill, Ontario	Hart (1947)[87]	Upper	5		42.1859	−1.6021	3	−0.9408	26.0	25.0		24.7
								10		31.0755	−1.0414	3	−0.8628	29.0	28.0		27.3
								15		20.8055	−0.6226	3	−0.9969	31.0	30.0		29.3
								20		21.0274	−0.5933	7	−0.9844	33.5	30.5		30.3
								25		16.8951	−0.4499	9	−0.9911	35.0	31.0		30.3
							Lower	20									0.7
								25									4.5
Semotilus atromaculatus (Creek chub)	Adult				Toronto, Ontario Knoxville, Tenn.	Hart (1952)[88]	Upper	10 (Toronto only)						29	28		27.5
								15 (Toronto only)		20.8055	−0.6226	3	−0.9969	31	30		29
								20 (Toronto only)		19.1315	−0.5328	6	−0.9856	33	30.5		30.5
								25		19.3186	−0.4717	18	−0.9921	36	32		31.5
								30		22.8982	−0.5844	19	−0.9961	37	33		31.5
Sphaeroides annulatus (Puffer)	Adult				Northern Gulf of Calif. Coast	Heath (1967)[89]	Upper	32.0		25.4649	−0.6088	3	−0.9716	37.0	36.0		
Sphaeroides maculatus (Northern puffer)		13.8–15.9 cm (average)	62.3–79.3 gm (average)	Mixed	New Jersey (40 N)	Hoff and Westman (1966)[90]	Upper	10		11.3999	−0.2821	3	−0.9988	30.0	25.0		27.5
								14		35.5191	−1.0751	3	−0.9449	32.0	27.0		30.2
								21		21.5353	−0.5746	3	−0.9814	32.0	30.0		31.2
								28		23.7582	−0.6183	3	−0.9239	33.5	31.1		32.5
							Lower	14		−1.7104	0.6141	4	0.9760	10.0	6.0		8.8
								21		−3.9939	0.7300	6	0.9310	12.0	8.0		10.7
								28		−7.4513	0.8498	5	0.9738	16.0	10.0		13.0
Thaleichthys pacificus (Eulachon or Columbia River Smelt)	Sexually Mature	161 mm ave.	31 gm ave.	Mixed	Cowlitz River, Wash.	Blahm & McConnell (1970)[100] unpublished data	Upper	5	river temp.	7.7440	−0.2740	7	−0.9142	29.0	8.0		10.5
Tilapia mossambica (Mozambique mouthbreeder)	4 months	8.0–12.0 cm	10.0–17.0 gm		Transvaal Africa	Allanson & Noble (1964)[71]	Upper	22		313.3830	−8.3878	4	−0.8898	37.10	36.5		36.94
								26		14.0458	−0.2800	5	−0.2140	37.92	37.5		37.7
								28		41.1610	−0.9950	4	−0.3107	38.09	37.9		37.89
								29		94.8243	−2.4125	5	−0.7781	38.10	37.0		37.91
								30		41.3233	−1.0018	6	−0.9724	38.50	37.6		37.59
								32		34.0769	−0.8123	4	−0.9209	38.4	37.6		37.6
								34		123.1504	−3.1223	3	−0.9938	38.4	38.2		38.25
								36		68.6764	−1.7094	6	−0.9053	38.77	37.9		38.2
Tinca tinca (tench)	Juvenile	4.6±0.4 cm		Mixed	England	Alabaster & Downing[69] (1966)	Upper	15		33.2000	1.0000	2[e]					
								20		29.6667	0.8333	3					
								25		27.1429	0.7143	2					

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[a] It is assumed in this table that the acclimation temperature reported is a true acclimation in the context of Brett (1952).[74]

[b] Number of median resistance times used for calculating regression equation.

[c] Correlation coefficient (perfect fit of all data points to the regression line = 1.0).

[d] = Incipient lethal temperature of Fry, et al., (1946).[83]

[e] See previous note for Alabaster 1967.[68]

APPENDIX II-D

Organochlorine Insecticides

Pesticide	Organism	Acute toxicity LC50		Sub-acute effects µg/liter	Reference
		µg/liter	hours		
ALDRIN	CRUSTACEANS				
	Gammarus lacustris	9800	96		Sanders 1969[124]
	Gammarus fasciatus	4300	96		Sanders in press[126]
	Palaemonetes kadiakensis	50	96		″
	Asellus brevicaudus	8	96		″
	Daphnia pulex	28	48		Sanders and Cope 1966[127]
	Simocephalus serrulatus	23	48		″
	INSECTS				
	Pteronarcys californica	1.3	96		Sanders and Cope 1968[129]
	Pteronarcys californica	180	96	2.5 µg/liter (30 day LC50)	Jensen and Gaufin 1966[113]
	Acroneuria pacifica	200	96	22 µg/liter (30 day LC50)	Jensen and Gaufin 1966[114]
	FISH				
	Pimephales promelas	28	96		Henderson et al. 1959[118]
	Lepomis macrochirus	13	96		″
	Salmo gairdneri	17.7	96		Katz 1961[119]
	Oncorhynchus kisutch	45.9	96		″
	Oncorhynchus tschawytscha	7.5	96		″
DDT	CRUSTACEAN				
	Gammarus lacustris	1.0	96		Sanders 1969[124]
	Gammarus fasciatus	0.8	96		Sanders in press[126]
	Palaemonetes kadiakensis	2.3	96		″
	Orconectes nais	0.24	96		″
	Asellus brevicaudus	4.0	96		″
	Simocephalus serrulatus	2.5	48		Sanders and Cope 1966[127]
	Daphnia pulex	0.36	48		″
	INSECT				
	Pteronarcys californica	7.0	96		Sanders and Cope 1968[129]
	Pteronarcella badia	1.9	96		″
	Claassenia sabulosa	3.5	96		″
	FISH				
	Pimephales promelas	19	96		Macek and McAllister 1970[131]
	Lepomis macrochirus	8	96		″
	Lepomis microlophus	5	96		″
	Micropterus salmoides	2	96		″
	Salmo gairdneri	7	96		″
	Salmo gairdneri			0.26 µg/l (15 day LC50)	FPRL Annual Report[137]
	Salmo trutta	2	96		Macek and McAllister 1970[131]
	Oncorhynchus kisutch	4	96		″
	Perca flavescens	9	96		″
	Ictalurus punctatus	16	96		″
	Ictalurus melas	5	96		″
TDE (DDD) Rhothane®	CRUSTACEAN				
	Gammarus lacustris	0.64	96		Sanders 1969[124]
	Gammarus fasciatus	0.86	96		Sanders in press[126]
	Palaemonetes kadiakensis	0.68	96		″
	Asellus breviacaudus	10.0	96		″
	Simocephalus serrulatus	4.5	48		Sanders and Cope 1966[127]
	Daphnia pulex	3.2	48		″
	INSECT				
	Pteronarcys californica	380	96		Sanders and Cope 1968[129]

Organochlorine Insecticides—Continued

Pesticide	Organism	Acute toxicity LC50		Sub-acute effects µg/liter	Reference
		µg/liter	hours		
DIELDRIN	CRUSTACEAN				
	Gammarus lacustris	460	96		Sanders 1969[124]
	Gammarus fasciatus	600	96		Sanders in press[126]
	Palaemonetes kadiakensis	20	96		"
	Orconectes nais	740	96		"
	Asellus brevicaudus	5	96		"
	Simocephalus serrulatus	190	48		Sanders and Cope 1966[127]
	Daphnia pulex	250	48		"
	INSECTS				
	Pteronarcys californica	0.5	96		Sanders and Cope 1968[128]
	Pteronarcys californica	39	96	2.0 (30 day LC50)	Jensen and Gaufin 1966[118]
	Acroneuria pacifica	24	96	0.2 (30 day LC50)	"
	Pteronarcella badia	0.5	96		Sanders and Cope 1968[128]
	Claassenia sabulosa	0.58	96		"
	FISH				
	Pimephales promelas	16	96		Henderson et al. 1959[113]
	Lepomis macrochirus	8	96		
	Salmo gairdneri	10	96		Katz 1961[119]
	Oncorhynchus kisutch	11	96		"
	Oncorhynchus tschawytscha	6	96		"
	Poecilia latipipna			3.0 (19 week LC50)	Lane and Livingston 1970[130]
	Poecilia latipipna			0.75 (reduced growth & reproduction—34 week)	"
	Lepomis gibbosus	6.7	96	1.7 (affect swimming ability and oxygen consumption—100-day)	Cairns and Scheir 1964[109]
	Ictaluras punctatus	4.5	96		FPRL[137]
CHLORDANE	CRUSTACEAN				
	Gammarus lacustris	26	96		Sanders 1969[124]
	Gammarus fasciatus	40	96		Sanders in press[126]
	Palaemonetes kadiakensis	4.0	96	2.5 (120 hour LC50)	"
	Simocephalus serrulatus	20	48		Sanders and Cope 1966[127]
	Daphnia pulex	29	48		"
	INSECT				
	Pteronarcys californica	15	96		Sanders and Cope 1968[128]
	FISH				
	Pimephales promelas	52	96		Henderson et al. 1959[113]
	Lepomis macrochirus	22	96		"
	Salmo gairdneri	44	96		Katz 1961[119]
	Oncorhynchus kisutch	56	96		"
	Oncorhynchus tschawytscha	57	96		"
ENDOSULFAN THIODAN	CRUSTACEAN				
	Gammarus fasciatus	5.8	96		Sanders 1969[124]
	Daphnia magna	52.9	96		Schoettger 1970[129]
	INSECT				
	Pteronarcys californica	2.3	96		Sanders and Cope 1968[128]
	Ischnura sp	71.8	96		Schoettger 1970[129]
	FISH				
	Salmo gairdneri	0.3	96		Schoettger 1970[129]
	Catastomus commersoni	3.0	96		"
ENDRIN	CRUSTACEAN				
	Gammarus lacustris	3.0	96		Sanders 1969[124]
	Gammarus fasciatus	0.9	120		Sanders in press[126]
	Palaemonetes kadiakensis	0.4	120		"
	Orconectes nais	3.2	96		"
	Asellus brevicaudus	1.5	96		"
	Simocephalus serrulatus	26	48		Sanders and Cope 1966[127]
	Daphnia pulex	20	48		"
	INSECT				
	Pteronarcys californica	0.25	96		Sanders and Cope 1968[128]
	Pteronarcys californica	2.4	96	1.2 (30 day LC50)	Jensen and Gaufin 1966[118]
	Acroneuria pacifica	0.32	96	0.03 (39 day LC50)	"
	Pteronarcella badia	0.54	96		Sanders and Cope 1968[128]
	Claassenia sabulosa	0.76	96		"
	FISH				
	Pimephales promelas	1.0	96		Henderson et al. 1959[113]
	Lepomis macrochirus	0.6	96		"
	Salmo gairdneri	0.6	96		Katz 1961[119]
	Oncorhynchus kisutch	0.5	96		"
	Oncorhynchus tschawytscha	1.2	96		"
HEPTACHLOR	CRUSTACEAN				
	Gammarus lacustris	29	96		Sanders 1969[124]
	Gammarus fasciatus	40	96		Sanders in press[126]
	Palaemonetes kadiakensis	1.8	96		"
	Orconectes nais	7.8	96		"
	Simocephalus serrulatus	47	48		Sanders and Cope 1966[127]
	Daphnia pulex	42	48		"

Organochlorine Insecticides—Continued

Pesticide	Organism	Acute toxicity LC50		Sub-acute effects µg/liter	Reference
		µg/liter	hours		
HEPTACHLOR	INSECTS				
	Pteronarcys californica	1.1	96		Sanders and Cope 1968[128]
	Pteronarcella badia	0.9	96		"
	Claassenia sabulosa	2.8	96		"
	FISH				
	Pimephales promelas	56	96		Henderson et al. 1959[113]
	Lepomis macrochirus	19	96		"
	Lepomis microlophus	17	96		Bridges 1961[107]
	Salmo gairdneri	19	96		Katz 1961[119]
	Oncorhynchus kisutch	59	96		"
	Oncorhynchus tschawytscha	17	96		"
LINDANE	CRUSTACEAN				
	Gammarus lacustris	48	96		Sanders 1969[124]
	Gammarus fasciatus	10	96		Sanders in press[126]
	Asellus brevicaudus	10	96		"
	Simocephalus serrulatus	520	48		Sanders and Cope 1966[127]
	Daphnia pulex	460	48		"
	INSECT				
	Pteronarcys californica	4.5	96		Sanders and Cope 1968[128]
	FISH				
	Pimephales promelas	87	96		Macek and McAllister 1970[121]
	Lepomis macrochirus	68	96		"
	Lepomis microlophus	83	96		"
	Micropterus salmoides	32	96		"
	Salmo gairdneri	27	96		"
	Salmo trutta	2	96		"
	Oncorhynchus kisutch	41	96		"
	Perca flavescens	68	96		"
	Ictalurus punctatus	44	96		"
	Ictalurus melas	64	96		"
METHOXYCHLOR	CRUSTACEAN				
	Gammarus lacustris	0.8	96		Sanders 1969[124]
	Gammarus fasciatus	1.9	96		Sanders in press[126]
	Palaemonetes kadiakensis	1.0	96		"
	Orconectes nais	0.5	96		"
	Asellus brevicaudus	3.2	96		"
	Simocephalus serrulatus	5	48		Sanders and Cope 1966[127]
	Daphnia pulex	0.78	48		"
	INSECT				
	Pteronarcys californica	1.4	96		Sanders and Cope 1968[128]
	Taeniopteryx nivalis	0.98	96		Merna unpublished data[130]
	Stenonema spp	0.63	96		Merna "
	FISH				
	Pimephales promelas	7.5	96	0.125 (reduced egg hatchability)	Merna unpublished data[130]
	Lepomis macrochirus	62.0	96		Henderson et al. 1959[113]
	Salmo gairdneri	62.6	96		Katz, 1961[119]
	Oncorhynchus kisutch	66.2	96		"
	Oncorhynchus tschawytscha	27.9	96		"
	Perca flavescens	20.0	96	0.6 (reduced growth) 8 months	Merna unpublished data[130]
TOXAPHENE	CRUSTACEAN				
	Gammarus lacustris	26	96		Sanders 1969[124]
	Gammarus fasciatus	6	96		Sanders in press[126]
	Palaemonetes kadiakensis	28	96		"
	Simocephalus serrulatus	10	48		Sanders and Cope 1966[127]
	Daphnia pulex	15	48		"
	INSECTS				
	Pteronarcys californica	2.3	96		Sanders and Cope 1968[128]
	Pteronarcella badia	3.0	96		"
	Claassenia sabulosa	1.3	96		"
	FISH				
	Pimephales promelas	14	96		Macek and McAllister 1970[121]
	Lepomis macrochirus	18	96		"
	Lepomis microlophus	13	96		"
	Micropterus salmoides	2	96		"
	Salmo gairdnerii	11	96		Macek and McAllister 1970[121]
	Salmo trutta	3	96		"
	Oncorhynchus kisutch	8	96		"
	Perca flavescens	12	96		"
	Ictalurus punctatus	13	96		"
	Ictalurus melas	5	96		"

Organophosphate Insecticides

Pesticide	Organism	Acute toxicity LC50		Sub-acute effects µg/liter	No effect µg/liter	Reference
		µg/liter	hours			
ABATE®	CRUSTACEAN					
	Gammarus lacustris	82	96			Sanders 1969[124]
	INSECT					
	Pteronarcys californica	10	96			Sanders and Cope 1968[128]
	FISH					
	Salmo gairdneri	158	96			FPRL[137]
AZINPHOSMETHYL GUTHION®	CRUSTACEANS					
	Gammarus lacustris	0.15	96			Sanders 1969[124]
	Gammarus fasciatus	0.10	96			Sanders in press[125]
	Gammarus pseudolimneaus				0.10-30 day	Bell unpublished data[134]
	Palaemonetes kadiakensis	1.2	120	0.16 (20 day LC50)		Sanders in press[126]
	Asellus brevicaudus	21.0	96			"
	INSECTS					
	Pteronarcys dorsata	12.1	96	4.9 (30 day LC50)		Bell unpublished data[134]
	Pteronarcys californica	1.5	96			Sanders and Cope 1968[128]
	Acroneuria lycorias			1.5 (30 day LC50)	1.36-30 day	Bell unpublished data[134]
	Ophiogomphus rupinsulensis	12.0	96	2.2 (30 day LC50)	1.73-30 day	"
	Hydropsyche bettoni			7.4 (30 day LC50)	4.94-30 day	"
	Ephemerella subvaria			4.5 (30 day LC50)	2.50-30 day	"
	FISH					
	Pimephales promelas	93	96			Katz 1961[119]
	Lepomis macrochirus	5.2	96			"
	Lepomis microlophus	52	96			Macek and McAllister 1970[131]
	Micropterus salmoides	5	96			"
	Salmo gairdneri	14	96			"
	Salmo trutta	4	96			"
	Oncorhynchus kisutch	17	96			"
	Perca flavescens	13	96			Macek and McAllister 1970[131]
	Ictalurus punctatus	3290	96			"
	Ictalurus melas	3500	96			"
AZINPHOSETHYL ETHYL GUTHION®	CRUSTACEANS					
	Simocephalus serrulatus	4	48			Sanders and Cope 1966[127]
	Daphnia pulex	3.2	48			"
	FISH					
	Salmo gairdneri	19	96			FPRL[137]
CARBOPHENOTHION TRITHION ®	CRUSTACEANS					
	Gammarus lactustris	5.2	96			Sanders 1969[124]
	Palaemonetes kadiakensis	1.2	96			Sanders in press[125]
	Asellus brevicaudus	1100	96			"
CHLOROTHION	CRUSTACEAN					
	Daphnia magna	4.5	48			*Water Qualiy Criteria* 1968
	FISH					
	Pimephales promelas	2800	96			Pickering et al. 1962[133]
	Lepomis macrochirus	700	96			"
CIODRIN®	CRUSTACEANS					
	Gammarus lacustris	15	96			Sanders 1965[124]
	Gammarus fasciatus	11	96			Sanders in press[126]
	FISH					
	Lepomis macrochirus	250	96			FPRL[137]
	Micropterus salmoides	1100	96			FPRL[137]
	Salmo gairdneri	55	96			FPRL[137]
	Ictalurus punctatus	2500	96			"
COUMAPHOS CO-RAL ®	CRUSTACEANS					
	Gammarus lacustris	0.07	96			Sarders 1969[124]
	Gammarus fasciatus	0.15	96			Sanders in press[126]
	Daphnia magna	1.0	48			*Water Quality Criteria* 1968
	INSECTS					
	Hydropsyche sp.	5	24			Carlson 1966[110]
	Hexagenia sp.	430	24			"
	FISH					
	Pimephales promelas	18000	96			Katz 1961[119]
	Lepomis macrochirus	180	96			"
	Salmo gairdneri	1500	96			"
	Oncorhynchus kisutch	15000	96			"
DEMETON SYSTOX®	CRUSTACEANS					
	Gammarus fasciatus	27	96			Sanders in press[126]
	FISH					
	Pimephales promelas	3200	96			Pickering et al. 1962[133]
	Lepomis macrochirus	100	96			"

Organophosphate Insecticides—Continued

Pesticide	Organism	Acute toxicity LC50		Sub-acute effects µg/liter	No effect µg/liter	Reference
		µg/liter	hours			
DIAZINON	**CRUSTACEANS**					
	Gammarus pseudolimneaus			0.27 (30 day LC50)	0.20 (30 day)	Bell unpublished data[134]
	Gammarus lacustris	200	96			Sanders 1969[124]
	Simocephalus serrulatus	1.4	48			Sanders and Cope 1966[127]
	Daphnia pulex	0.90	48			"
	Daphnia magna				0.26 (21 day)	Biesinger unpublished data[135]
	INSECTS					
	Pteronarcys californica	25	96			Sanders and Cope 1968[128]
	Pteronarcys dorsata			4.6 (30 day LC50)	3.29 (30 day)	Bell unpublished data[134]
	Acroneuria lycorias	1.7	96	1.25 (30 day LC50)	0.83 (30 day)	"
	Ophiogomphus rupinsulensis			2.2 "	1.29 "	"
	Hydropsyche bettoni			3.54 "	1.79 "	"
	Ephemerella subvaria			1.05 "	0.42 "	"
DICHLORVOS DDVP VAPONA®	**CRUSTACEANS**					
	Gammarus lacustris	0.50	96			Sanders 1969[124]
	Gammarus faciatus	0.40	96			Sanders in press[126]
	Simocephalus serrulatus	0.26	48			Sanders and Cope 1966[127]
	Daphnia pulex	0.07	48			"
	INSECTS					
	Pteronarcys californica	0.10	96			Sanders and Cope 1968[128]
	FISH					
	Lepomis macrochirus	869	96			FPRL[137]
DIOXATHION DELNAV®	**CRUSTACEANS**					
	Gammarus lacustris	270	96			Sanders 1969[124]
	Gammarus fasciatus	8.6	96			Sanders in press[126]
	FISH					
	Pimephales promelas	9300	96			Pickering et al. 1962[128]
	Lepomis macrochirus	34	96			"
	Lepomis cyanellus	61	96			"
	Micropterus salmoides	36	96			"
DISULFOTON DI-SYSTON®	**CRUSTACEANS**					
	Gammarus lacustris	52	96			Sanders 1969[124]
	Gammarus fasciatus	21	96			Sanders in press[126]
	Palaemonetes kadiakensis	38	96			"
	INSECTS					
	Pteronarcys californica	5	96			Sanders and Cope 1968[128]
	Pteronarcys californica	24	96	1.9 (30 day LC50)		Jensen and Gaufin 1964[117]
	Acroneuria pacifica	8.2	96	1.4 (30 day LC50)		"
	FISH					
	Pimephales promelas	3700	96			Pickering et al. 1962[128]
	Lepomis macrochirus	63	96			"
DURSBAN®	**CRUSTACEANS**					
	Gammarus lacustris	0.11	96			Sanders 1969[124]
	Gammarus fasciatus	0.32	96			Sanders in press[126]
	INSECTS					
	Pteronarcys californica	10	96			Sanders and Cope 1968[128]
	Pteronarcella badia	0.38	96			"
	Claassenia sabulosa	0.57	96			"
	FISH					
	Lepomis macrochirus	2.6	96			FPRL[137]
	Salmo gairdneri	11	96			FPRL[137]
ETHION NIALATE®	**CRUSTACEANS**					
	Gammarus lacustris	1.8	96			Sanders 1969[124]
	Gammarus fasciatus	9.4	96			Sanders in press[126]
	Palaemonetes kadiakensis	5.7	96			Sanders in press[126]
	INSECTS					
	Pteronarcys californica	2.8	96			Sanders and Cope 1968[128]
	FISH					
	Lepomis macrochirus	220	96			FPRL[137]
	Micropterus salmoides	150	96			"
	Salmo gairdneri	560	96			"
	Salmo clarkii	720	96			"
	Ictalurus punctatus	7500	96			"
EPN	**CRUSTACEAN**					
	Gammarus lacustris	15	96			Sanders 1969[124]
	Gammarus fasciatus	7	96			Sanders in press[126]
	Palaemonetes kadiakensis	0.56	96			"
	FISH					
	Pimephales promelas	110000	96			Solon and Nair 1970[130]
	Lepomis macrochirus	100	96			Pickering et al. 1962[128]

Organophosphate Insecticides—Continued

Pesticide	Organism	Acute toxicity LC50 µg/liter	Acute toxicity LC50 hours	Sub-acute effects µg/liter	No effect µg/liter	Reference
FENTHION BAYTEX®	CRUSTACEANS					
	Gammarus lacustris	8.4	96			Sanders 1969[124]
	Gammarus fasciatus	110	96			Sanders in press[126]
	Palaemonetes kadiakensis	5	120	1.5 (20 day LC50)		"
	Orconectes nais	50	96			"
	Asellus brevicaudus	1800	96			"
	Simocephalus serrulatus	0.62	48			Sanders and Cope 1966[127]
	Daphnia pulex	0.80	48			"
	INSECTS					
	Pteronarcys californica	4.5	96			Sanders and Cope 1968[128]
	FISH					
	Pimephales promelas	2440	96			Macek and McAllister 1970[131]
	Lepomis macrochirus	1380	96			"
	Lepomis microlophus	1880	96			"
	Micropterus salmoides	1540	96			"
	Salmo gairdneri	930	96			"
	Salmo trutta	1330	96			"
	Oncorhynchus kisutch	1320	96			"
	Perca flavesens	1650	96			"
	Ictalurus punctatus	1680	96			"
	Ictalurus melas	1620	96			"
MALATHION	CRUSTACEANS					
	Gammarus pseudolimneaus			0.023 (30 day LC50)	0.008–30 day	Bell unpublished data[134]
	Gammarus lacustris	1.0	96			Sanders 1969[124]
	Gammarus fasciatus	0.76	96	0.5 (120 hour LC50)		Sanders in press[126]
	Palaemonetes kadiakensis	12	96	9.0 "		"
	Orconectes nais	180	96			"
	Asellus brevicaudus	3000	96			"
	Simocephalus serrulatus	3.5	48			Sanders and Cope 1966[127]
	Daphnia pulex	1.8	48			"
	Daphnia magna				0.6–21 day	Biesinger unpublished data[135]
	INSECTS					
	Pteronarcys californica	10	96			Sanders and Cope 1968[128]
	Pteronarcys dorsata			11.1 (30 day LC50)	9.4–30 day	Bell unpublished data[134]
	Acroneuria lycorias	1.0		0.3 (30 day LC50)	0.17–30 day	"
	Pteronarcella badia	1.1	96			Sanders and Cope 1968[128]
	Classenia sabulosa	2.8	96			"
	Boyeria vinosa			2.3 (30 day LC50)	1.65–30 day	Bell unpublished data[134]
	Ophiogomphus rupinsulensis			0.52 "	0.28–30 day	"
	Hydropsyche bettoni			0.34 "	0.24–30 day	"
	FISH					
	Pimephales promelas	9000	96	580 (spinal deformity 10 month)	200–10 month exposure	Mount and Stephen 1967[122]
	Lepomis macrochirus	110	96	7.4 (spinal deformity several months)	3.6–11 months	Eaton 1971[111]
	Lepomis cyanellus	120	96			Pickering et al. 1962[123]
	Lepomis microlophus	170	96			Macek and McAllister 1970[131]
	Micropterus salmoides	285	96			"
	Salmo gairdneri	170	96			"
	Salmo trutta	200	96			"
	Oncorhynchus kisutch	101	96			"
	Perca flavescens	263	96			"
	Ictalurus punctatus	8970	96			"
	Ictalurus melas	12900	95			"
METHYL PARATHION BAYER E601	FISH					
	Pimephales promelas	8900	96			Macek and McAllister 1970[131]
	Lepomis macrochirus	5720	96			"
	Lepomis microlophus	5170	96			"
	Micropterus salmoides	5220	96			"
	Salmo gairdneri	2750	96			"
	Salmo trutta	4740	96			"
	Oncorhynchus kisutch	5300	96			"
	Perca flavescens	3060	96			"
	Italurus punctatus	5710	96			"
	Italurus melas	6640	96			"
MEVINPHOS PHOSDRIN®	CRUSTACEAN					
	Gammarus lacustris	130	96			Sanders 1969[124]
	Gammarus fasciatus	2.8	96			Sanders in press[126]
	Palaemonetes kadiakensis	12	96			"
	Asellus brevicandus	56	96			"
	Simocephalus serrulatus	0.43	48			Sanders and Cope 1966[127]
	Daphnia pulex	0.16	48			"

Organophosphate Insecticides—Continued

Pesticide	Organism	Acute toxicity LC50		Sub-acute effects µg/liter	No effect µg/liter	Reference
		µg/liter	hours			
MEVINPHOS PHOSDRIN®	INSECTS					
	Pteronarcys californica	5.0	96			Sanders and Cope 1968[128]
	FISH					
	Lepomis macrochirus	70	96			FPRL[137]
	Micropterus salmoides	110	96			FPRL[137]
NALED DI BROM®	CRUSTACEANS					
	Gammarus lacustris	110	96			Sanders 1969[124]
	Gammarus fasciatus	14	96			Sanders in press[126]
	Palaemonetes kadiakensis	90	96			"
	Orconectes nais	1800	96			"
	Asellus brevicaudus	230	96			"
	Simocephalus serrulatus	1.1	48			Sanders and Cope 1966[127]
	Daphnia pulex	0.35	48			"
	INSECTS					
	Pteronarcys californica	8.0	96			Sanders and Cope 1968[128]
	FISH					
	Lepomis macrochirus	180	96			FPRL[137]
	Salmo gairdneri	132	96			FPRL[137]
OXYDEMETON METHYL META-SYSTOX®	CRUSTACEANS					
	Gammarus lacustris	190	96			Sanders 1969[124]
	Gammarus fasciatus	1000	96			Sanders in press[126]
	INSECTS					
	Pteronarcys californica	35	96			Sanders and Cope 1968[128]
	FISH					
	Lepomis macrochirus	14000	96			FPRL[137]
	Salmo gairdneri	4000	96			FPRL[137]
PARATHION	CRUSTACEANS					
	Gammarus lacustris	3.5	96			Sanders 1969[124]
	Gammarus fasciatus	2.1	96	1.6 (120 hour LC50)		Sanders in press[126]
	Palaemonetes kadiakensis	1.5	96			"
	Simocephalus sarrulatus	0.37	48			Sanders and Cope 1966[127]
	Daphnia pulex	0.60	48			"
	Orconectes nais	0.04	96			Sanders in press[126]
	Asellus brevicaudus	600	96			"
	INSECTS					
	Pteronarcys californica	36	96	2.2 (30 day LC50)		Jensen and Gaufin 1964[117]
	Pteronarcys dorsata	3.0	96	0.90 (30 day LC50)		Bell unpublished data[134]
	Pteronarcella badia	4.2	96			Sanders and Cope 1968[128]
	Claassenia sabulosa	1.5	96			"
	Acroneuria pacifica	3.0	96	0.44 (30 day LC50)		Jensen and Gaufin 1964[117]
	Acroneuria lycorias			0.013 (30 day LC50)		Bell unpublished data[134]
	Ephemerella subvaria	0.16	96	0.056 (30 day LC50)		Bell unpublished data[134]
	Ophigomphus rupinsulensis	3.25	96	0.22 "		"
	Hydropsyche bettoni			0.45 "		"

Organophosphate Insecticides—Continued

Pesticide	Organism	Acute toxicity LC50		Sub-acute effects µg/liter	No effect µg/liter	Reference
		µg/liter	hours			
PARATHION	FISH					
	Pimephales promelas	1410	96			Solon and Nair 1970[130]
	Lepomis macrochirus	65	96			Pickering et al. 1962[123]
	Lepomis cyanellus	425	96			″
	Micropterus salmoides	190	96			″
PHORATE THIMET®	CRUSTACEANS					
	Gammarus lacustris	9	96			Sanders 1969[134]
	Gammarus fasciatus	0.60	96			Sanders in press[136]
	Orconectes nais	50	96			″
PHOSPHAMIDON	CRUSTACEANS					
	Gammarus lacustris	2.8	96			Sanders 1969[134]
	Gammarus fasciatus	16	96			Sanders in press[136]
	Orconectes nais	7500	96			″
	Simocephalus serrulatus	6.6	48			Sanders and Cope 1966[137]
	Daphnia pulex	8.8	48			″
	INSECTS					
	Pteronarcys californica	150	96			Sanders and Cope 1968[138]
	FISH					
	Pimephales promelas	100000	96			FPRL[137]
	Lepomis macrochirus	4500	96			″
	Ictalurus punctatus	70000	96			″
RONNEL	FISH					
	Pimephales promelas	305	96			Solon and Nair 1970[130]
T EPP	CRUSTACEANS					
	Gammarus lacustris	38	96			Sanders 1969[134]
	Gammarus fasciatus	210	96			Sanders in press[136]
	FISH					
	Pimephales promelas	1900	96			Pickering et al. 1962[123]
	Lepomis macrochirus	1100	96			″
TRICHLOROPHON DIPTEREX DYLOX	CRUSTACEANS					
	Gammarus lacustris	40	96			Sanders 1969[134]
	Simocephalus serrulatus	0.32	48			Sanders and Cope 1966[136]
	Daphnia pulex	0.18	48			″
	INSECTS					
	Pteronarcys californica	69	96	9.8 (30 day LC50)		Jensen and Gaufin 1964[117]
	Pteronarcys californica	35	96			Sanders and Cope 1968[128]
	Acroneuria pacifica	16.5	96	8.7 (30 day LC50)		Jensen and Gaufin 1964[117]
	Pteronarcella badia	11	96			Sanders and Cope 1968[128]
	Claassenia sabulosa	22	96			Sanders and Cope 1968[128]
	FISH					
	Pimephales promelas	109000	96			Pickering et al. 1962[123]
	Lepomis macrochirus	3800	96			″

Carbamate

Pesticide	Organism	Acute toxicity LC50		Sub-acute effects µg/liter	No effect µg/liter	Reference
		µg/liter	hours			
CARBARYL SEVIN®	CRUSTACEANS					
	Gammarus lacustris	16	96			Sanders 1969[124]
	Gammarus fasciatus	26	96			Sanders in press[126]
	Palaemonetes kadiakensis	5.6	96			"
	Orconectes nais	8.6	96			"
	Asellus brevicaudus	240	96			"
	Simocephalus serrulatus	7.6	48			Sanders and Cope 1966[127]
	Daphnia pulex	6.4	48			"
	Daphnia magna				5.0 63 day	Biesinger unpublished data[135]
	INSECTS					
	Pteronarcys californica	4.8	96			Sanders and Cope 1968[128]
	Pteronarcys dorsata			23.0 (30 day LC50)	11.5 30 day	Bell unpublished data[134]
	Pteronarcella badia	1.7	96			Sanders and Cope 1968[128]
	Claassenia sabulosa	5.6	96			"
	Acroneuria lycorias			2.2 (30 day LC50)	1.3 30 day	Bell unpublished data[134]
	Hydropsyche bettoni			2.7 (30 day LC50)	1.8 30 day	"
	FISH					
	Pimephales promelas	9000	96	680 (decline survival and reproduction 6 months)	210 (6 month)	Carlson unpublished data[136]
	Lepomis macrochirus	6760	96			Macek and McAllister 1970[131]
	Lepomis microlophus	11200	96			"
	Micropterus salmoides	6400	96			"
	Salmo gairdneri	4340	96			"
	Salmo trutta	1950	96			"
	Oncorhynchus kisutch	764	96			"
	Perca flavescens	745	96			"
	Ictalurus punctatus	15800	96			"
	Ictalurus melas	20000	96			"
BAYGON	CRUSTACEANS					
	Gammarus lacustris	34	96			Sanders 1969[124]
	Gammarus fasciatus	50	96			Sanders in press[126]
	INSECT					
	Pteronarcys californica	13	96			Sanders and Cope 1968[128]
AMINOCARB METACIL	CRUSTACEAN					
	Gammarus lacustris	12	96			Sanders 1969[124]
BAYER 37344	INSECTS					
	Pteronarcys californica	5.4	96			Sanders and Cope 1968[128]
ZECTRAN	CRUSTACEANS					
	Gammarus lacustris	46	96			Sanders 1969[124]
	Gammarus fasciatus	40	96			Sanders in press[126]
	Palaemonetes kadiakensis	83	96	25 (20 day LC50)		"
	Simocephalus serrulatus	13	48			Sanders and Cope 1966[127]
	Daphnia pulex	10	48			"
	INSECTS					
	Pteronarcys californica	10	96			Sanders and Cope 1968[128]
	FISH					
	Pimephales promelas	17000	96			Macek and McAllister 1970[131]
	Lepomis macrochirus	11200	96			"
	Lepomis microlophus	16700	96			"
	Micropterus salmoides	14700	96			"
	Salmo gairdneri	10200	96			"
	Salmo trutta	8100	96			"
	Oncorhynchus kisutch	1730	96			"
	Perca flavescens	2480	96			"
	Ictalurus punctatus	11400	96			"
	Ictalurus melas	16700	96			"

Herbicides, Fungicides, Defoliants

Pesticide	Organism	Acute toxicity LC50		Sub-acute effects µg/liter	No effect µg/liter	Reference
		µg/liter	hours			
ACROLEIN AQUALIN	**FISH**					
	Lepomis macrochirus	80	24			Bond et al. 1960[106]
	Salmo trutta	46	24			Burdick et al. 1964[108]
	Lepomis macrochirus	79	24			"
AMINOTRIAZOLE AMITROL	**CRUSTACEAN**					
	Gammarus fasciatus				100,000 µg/l 48 hr.	Sanders 1970[125]
	Daphnia magna	30000	48			"
	Cypridopsis vidua	32060	48			"
	Asellus brevicaudus					"
	Palaemonetes kadiakensis				100,000 µg/l 48 hr.	"
	Orconectes nais				100,000 µg/l 48 hr.	"
	FISH					
	Lepomis macrochirus				100,000 µg/l 48 hr.	Sanders 1970[125]
	Oncorhyncus kisutch	325000	48			Bond et al. 1960[106]
BALAN	**CRUSTACEAN**					
	Gammarus faciatus	1100	96			Sanders 1970[125]
BENSULFIDE	**CRUSTACEAN**					
	Gammarus faciatus	1400	96			Sanders 1970[125]
CHLOROXURON	**FISH**					
	Lepomis macrochirus	25000	48			Hughes and Davis 1964[116]
CIPC	**FISH**					
	Lepomis macrochirus	8000	48			Hughes and Davis 1964[116]
DACTHAL	**FISH**					
	Lepomis macrochirus	700000	48			Hughes and Davis 1964[116]
DALAPON (SODIUM SALT)	**CRUSTACEAN**					
	Simocephalus serrulatus	16000	48			Sanders and Cope 1966[127]
	Daphnia pulex	11000	48			"
	INSECT					
	Pteronarcys californica				100,000 µg/l 96 hr.	Sanders and Cope 1968[128]
	FISH					
	Pimephales promelas	290000	96			Surber and Pickering 1962[131]
	Lepomis macrochirus	290000	96			"
	Oncorhynchus kisutch	340000	48			Bond et al. 1960[106]
DEF	**CRUSTACEAN**					
	Gammarus lacustris	100	96			Sanders 1969[124]
	INSECT					
	Pteronarcys californica	2100	96			Sanders and Cope 1968[128]
DEXON	**CRUSTACEAN**					
	Gammarus lacustris	3700	96			Sanders 1969[124]
	INSECT					
	Pteronarcys californica	24000	96			Sanders and Cope 1968[128]
DICAMBA	**CRUSTACEAN**					
	Gammarus lacustris	3900	96			Sanders 1969[124]
	Gammarus fasciatus				100,000 µg/l 48 hr.	Sanders 1970[125]
	Daphnia magna				100,000 µg/l 48 hr.	"
	Cypridopsis vidua				100,000 µg/l 48 hr.	"
	Asellus brevicaudus				100,000 µg/l 48 hr.	"
	Palaemonetes kadiakensis				100,000 µg/l 48 hr.	'
	Orconectes nais				100,000 µg/l 48 hr.	"
	FISH					
	Lepomis macrochirus	20000	48			Hughes and Davis 1962[114]
DICHLOBENIL CASARON®	**CRUSTACEAN**					
	Gammarus lacustris	11000	96			Sanders 1969[124]
	Gammarus fasciatus	10000	96			Sanders 1970[125]
	Hyallella azteca	8500	96			Wilson and Bond 1969[133]
	Simocephalus serrulatus	5800	48			Sanders and Cope 1968[128]
	Daphnia pulex	3700	48			"
	Daphnia magna	10000	48			Sanders 1970[125]
	Cypridopsis vidua	7800	48			"
	Asellus brevicaudus	34600	48			"
	Palaemonetes kadiakensis	9000	48			"
	Orconectes nais	22000	48			"
	INSECTS					
	Pteronarcys californica	7000	96			Sanders and Cope 1968[128]
	Tendipedidae	7800	96			Wilson and Bond 1969[133]
	Callibaetes sp.	10300	96			"
	Limnephilus	13000	96			"
	Enallegma	20700	96			"
	FISH					
	Lepomis macrochirus	20000	48			

Herbicides, Fungicides, Defoliants—Continued

Pesticide	Organism	Acute toxicity LC50		Sub-acute effects µg/liter	No effect µg/liter	Reference
		µg/liter	hours			
DICHLONE PHYGON XL	**CRUSTACEAN**					
	Gammarus lacustris	1100	96			Sanders 1969[124]
	Gammarus fasciatus	100	96			Sanders 1970[125]
	Daphnia magna	25	48			"
	Cypridopsis vidua	120	48			"
	Asellus brevicaudus	200	48			"
	Palaemonetes kadiakensis	450	48			"
	Orconectes nais	3200	48			"
	FISH					
	Lepomis macrochirus	70	48			Bond et al. 1960[106]
	Micropterus salmoides	120	48			Hughes and Davis 1962[114]
DIQUAT	**CRUSTACEAN**					
	Hyalella azteca	48	96			Wilson and Bond 1969[133]
	INSECTS					
	Callibaetes sp.	16400	96			Wilson and Bond 1969[133]
	Limnephilus	33000	96			"
	Tendipedidae	>100000	96			"
	Enallagma	>100000	96			"
	FISH					
	Pimephales promelas	14000	96			Surber and Pickering 1962[131]
	Lepomis macrochirus	35000	96			Gilderhus 1967[112]
	Micropterus salmoides	7800	96			Surber and Pickering 1962[131]
	Esox lucius	16000	48			Gilderhus 1967[112]
	Stizostedion vitreum vitreum	2100	96			"
	Salmo gairdneri	11200	48			"
	Oncorhynchus tshawytscha	28500	48			Bond et al. 1960[106]
DIURON	**CRUSTACEAN**					
	Gammarus lacustris	160	96			Sanders 1969[124]
	Gammarus fasciatus	700	96			Sanders 1970[125]
	Simocephalus serrulatus	2000	48			Sanders and Cope 1966[127]
	Daphnia pulex	1400	48			"
	INSECT					
	Pteronarcys californica	1200	96			Sanders and Cope 1968[128]
	FISH					
	Oncorhynchus kisutch	16000	48			Bond et al. 1960[106]
DIFOLITAN	**CRUSTACEAN**					
	Gammarus lacustris	800	96			Sanders 1969[124]
	INSECT					
	Pteronarcys californica	40	96			Sanders and Cope 1968[128]
DINITROBUTYL PHENOL	**CRUSTACEAN**					
	Gammarus fasciatus	1800	96			Sanders 1970[125]
DIPHENAMID	**CRUSTACEAN**					
	Gammarus fasciatus				100,000 µg/l 48 hr.	Sanders 1970[125]
	Daphnia magna	58000	48			"
	Cypridopsis vidua	50000	48			"
	Asellus brevicaudus				100,000 µg/l 48 hr.	"
	Palaemonetes kadiakensis	58000	48			"
	Orconectes nais				100,000 µg/l 48 hr.	"
DURSBAN	**CRUSTACEAN**					
	Gammarus lacustris	0.11	96			Sanders 1969[124]
	INSECT					
	Pteronarcys californica	10	96			Sanders and Cope 1968[128]
	Pteronarcella badia	0.38	96			"
	Claassenia sabulosa	0.57	96			"
2,4-D (PGBE)	**CRUSTACEAN**					
	Gammarus lacustris	1600	96			Sanders 1969[124]
	Gammarus fasciatus	2500	96			Sanders 1970[125]
	Daphnia magna	100	48			"
	Cypridopsis vidua	320	48			"
	Asellus brevicaudus	2200	48			"
	Palaemonetes kadiakensis	2700	48			"
	Orconectes nais				100,000 µg/l 48 hr.	"
2,4-D (BEE)	**CRUSTACEAN**					
	Gammarus lacustris	440	96			Sanders 1969[124]
	Gammarus fasciatus	5900	96			Sanders 1970[125]
	Daphnia magna	5600	48			"
	Cypridopsis vidua	1800	48			"
	Asellus brevicaudus	3200	48			"
	Palaemonetes kadiakensis	1400	48			"
	Orconectes nais				100,000 µg/l 48-hr	"
	INSECT					
	Pteronarcys californica	1600	96			Sanders and Cope 1968[128]

Herbicides, Fungicides, Defoliants—Continued

Pesticide	Organism	Acute toxicity LC50		Sub-acute effects µg/liter	No effect µg/liter	Reference
		µg/liter	hours			
2,4 D (BEE)	FISH					
	Pimephales promelas	5600	96	1500 µg/l lethal to eggs in 48 hour exposure	300 µg/l 10 mo.	Mount and Stephan 1967[123]
2,4-D (IOE)	CRUSTACEAN					
	Gammarus lacustris	2400	96			Sanders 1969[174]
2,4-D (DIETHYLAMINE SALT)	CRUSTACEAN					
	Gammarus lacustris					Sanders 1969[124]
	Gammarus fasciatus				100,000 µg/l 48 hr.	Sanders 1970[125]
	Daphnia magna	4000	48			"
	Crypidopsis vidua	8000	48			"
	Asellus brevicaudus				100,000 µg/l 48 hr.	"
	Palaemonetes kadiakensis				100,000 µg/l 48 hr.	"
	Orconectes nais				100,000 µg/l 48 hr.	"
ENDOTHALL DI SODIUM SALT	FISH					
	Pimephales notatus	110000	96			Walker 1964[132]
	Lepomis macrochirus	125000	96			"
	Micropterus salmoides	120000	96			"
	Notropis umbratilus	95000	96			"
	Micropterus salmoides	200000	96			Bond et al. 1960[106]
	Oncorhynchus tschawytscha	136000	96			"
ENDOTHALL DIPOTASSIUM SALT	CRUSTACEAN					
	Gammarus lacustris				100,000 µg/l 96 hr.	Sanders 1969[124]
	FISH					
	Pimephales promelas	320000	96			Surber and Pickering 1962[131]
	Lepomis macrochirus	160000	96			"
EPTAM	CRUSTACEAN					
	Gammarus fasciatus	23000	96			Sanders 1970[126]
FENAC (SODIUM SALT)	CRUSTACEAN					
	Gammarus lacustris	12000	96			Sanders 1969[124]
	Gammarus fasciatus				100,000 µg/l 48 hr.	Sanders 1970[125]
	Daphnia pulex	4500	48			Sanders and Cope 1966[127]
	Simocephalus serrulatus	6600	48			"
	Daphnia magna				100,000 µg/l 48 hr.	Sanders 1970[125]
	Cypridopsis vidua				100,000 µg/l 48 hr.	"
	Asellus brevicaudus				100,000 µg/l 48 hr.	"
	Palaemonetes kadiakensis				100,000 µg/l 48 hr.	"
	Orconectes nais					
	INSECT					
	Pteronarcys californica	55000	96			Sanders and Cope 1968[128]
	FISH					
	Lepomis	15000	48			Hughes and Davis 1962[114]
HYAMINE 1622	FISH					
	Pimephales promelas	1600	96			Surber and Pickering 1962[131]
	Lepomis macrochirus	1400	96			"
	Oncorhynchus kisutch	53000	96			Bond et al. 1960[106]
HYAMINE 2389	FISH					
	Pimephales promelas	2400	96			Surber and Pickering 1962[131]
	Lepomis macrochirus	1200	96			"
HYDROTHAL 47	CRUSTACEAN					
	Gammarus fasciatus	510	96			Sanders 1970[126]
HYDROTHAL 191	CRUSTACEAN					
	Gammarus lacustris	500	96			Sanders 1969[124]
	Gammarus fasciatus	480	96			Sanders 1970[125]
HYDROTHAL PLUS	FISH					
	Lepomis macrochirus	3500	48			Hughes and Davis 1964[116]
IPC	CRUSTACEAN					
	Gammarus lacustris	10000	96			Sanders 1969[124]
	Gammarus fasciatus	19000	96			Sanders 1970[125]
	Simocephalus serrulatus	10000	48			Sanders and Cope 1966[137]
	Daphnia pulex	10000	48			"
KURON	CRUSTACEAN					
	Simocephalus serrulatus	2400	48			Sanders and Cope 1966[137]
	Daphnia pulex	2000	48			"
MCPA	FISH					
	Lepomis macrochirus	1500	48			Hughes and Davis 1964[116]
MOLINATE	CRUSTACEAN					
	Gammarus lacustris	4500	96			Sanders 1969[134]
	Gammarus fasciatus	300	96			Sanders 1970[126]
	Daphnia magna	600	48			"
	Asellus brevicaudus	400	48			"
	Palaemonetes kadiakensis	1000	48			"
	Orconectes nais	5600	48			"

Herbicides, Fungicides, Defoliants—Continued

Pesticide	Organism	Acute toxicity LC50		Sub-acute effects µg/liter	No effect µg/liter	Reference
		µg/liter	hours			
MONURON	FISH					
	Oncorhynchus kisutch	110000	48			Bond et al. 1960[106]
PARAQUAT	CRUSTACEAN					
	Gammarus lacustris	11000	96			Sanders 1969[124]
	Simocephalus serrulatus	4000	48			Sanders and Cope 1966[127]
	Daphnia pulex	3700	48			"
	INSECT					
	Pteronarcys californica				100,000 µg/l 96 hr.	Sanders and Cope 1968[128]
PEBULATE	CRUSTACEAN					
	Gammarus fasciatus	10000	96			Sanders 1970[125]
PICLORAM	CRUSTACEAN					
	Gammarus lacustris	27000	96			Sanders 1969[124]
	INSECT					
	Pteronarcys californica	48000	96			Sanders and Cope 1968[128]
PROPANIL	CRUSTACEAN					
	Gammarus fasciatus	16000	96			Sanders 1970[125]
SILVEX (BEE)	CRUSTACEAN					
	Gammarus fasciatus	250	96			Sanders 1970[125]
	Daphnia magna	2100	48			"
	Cypridopsis vidua	4900	48			"
	Asellus brevicaudus	40000	48			"
	Palaemonetes kadiakensis	8000	48			"
	Orconectes nais	60000	48			"
	FISH					
	Lepomis macrochirus	1100	48			Hughes and Davis 1963[115]
SILVEX (PGBE)	CRUSTACEAN					
	Gammarus fasciatus	840	96			Sanders 1970[125]
	Daphnia magna	180	48			"
	Cypridopsis vidua	200	48			"
	Asellus brevicaudus	500	48			"
	Palaemonetes kadiakensis	3200	48			"
	Orconectes nais				100,000 µg/l 48 hr.	"
	FISH					
	Lepomis macrochirus	16600	48			Hughes and Davis 1963[115]
SILVEX (IOE)	FISH					
	Lepomis macrochirus	10000	48			Hughes and Davis 1963[115]
SILVEX (POTASSIUM SALT)	FISH					
	Lepomis macrochirus	83000	48			Hughes and Davis 1963[115]
SIMAZINE	CRUSTACEAN					
	Gammarus lacustris	13000	96			Sanders 1969[124]
	Gammarus fasciatus				100,000 µg/l 48 hr.	Sanders 1970[125]
	Daphnia magna	1000	48			
	Cypridopsis vidua	3200	48			
	Asellus brevicaudus				100,000 µg/l 48 hr.	Sanders 1970[125]
	Palaemonetes kadiakensis				100,000 µg/l 48 hr.	"
	Orconectes nais				100,000 µg/l 48 hr.	"
	FISH					
	Oncorhynchus kisutch	6600	48			Bond et al. 1960[106]
TRIFLURALIN	CRUSTACEAN					
	Gammarus lacustris	2200	96			Sanders 1969[124]
	Gammarus fasciatus	1000	96			Sanders 1970[125]
	Daphnia magna	560	48			"
	Daphnia pulex	240	48			Sanders and Cope 1966[127]
	Simocephalus serrulatus	450	48			"
	Cypridopsis vidua	250	48			Sanders 1970[125]
	Asellus brevicaudus	200	48			"
	Palaemonetes kadiakensis	1200	48			"
	Orconectes nais	50000	48			"
	INSECT					
	Pteronarcys californica	3000	96			Sanders and Cope 1968[128]
VERNOLATE	CRUSTACEAN					
	Gammarus lacustris	1800	96			Sanders 1969[124]
	Gammarus fasciatus	13000	96			Sanders 1970[125]
	Daphnia magna	1100	48			"
	Cypridopsis vidua	240	48			"
	Asellus bravicaudus	5600	48			"
	Palaemonetes kadiakensis	1900	48			"
	Orconectes nais	24000	48			"

Botanicals

Pesticide	Organism	Acute toxicity LC50		Sub-acute effects µg/liter	No effect µg/liter	Reference
		µg/liter	hours			
ALLETHRIN	**CRUSTACEAN**					
	Gammarus lacustris	11	96			Sanders 1969[124]
	Gammarus fasciatus	8	96			Sanders in press[126]
	Simocephalus serrulatus	56	48			Sanders and Cope 1966[127]
	Daphnia pulex	21	48			"
	INSECTS					
	Pteronarcys californica	2.1	96			Sanders and Cope 1968[128]
	FISH					
	Lepomis macrochirus	56	96			FPRL[137]
	Salmo gairdneri	19	96			"
PYRETHRUM	**CRUSTACEANS**					
	Gammarus lacustris	12	96			Sanders 1969[124]
	Gammarus fasciatus	11	96			"
	Simocephalus serrulatus	42	48			Sanders and Cope 1966[127]
	Daphnia pulex	25	48			"
	INSECTS					
	Pteronarcys californica	1.0	96			Sanders and Cope 1968[128]
ROTENONE	**CRUSTACEANS**					
	Gammarus lacustris	2600	96			Sanders 1969[illegible]
	Simocephalus serrulatus	190	48			Sanders and Cope 1966[127]
	Daphnia pulex	100	48			"
	INSECTS					
	Pteronarcys californica	380	96			Sanders and Cope 1968[128]

APPENDIX II-E

GUIDELINES FOR AQUATIC TOXICOLOGICAL RESEARCH ON PESTICIDES

More than one billion pounds of pesticides were produced in the United States in 1969 (Fowler et al. 1971).[152] However, before such materials can be transported in interstate commerce, they must be registed according to provisions of the Federal Insecticide, Fungicide, and Rodenticide Act and amendments. Responsibility for implementing this act is vested in the Pesticide Regulation Division of the Environmental Protection Agency. Properties of pesticides that must be considered in the registration process include: efficacy on the intended pest; safety to the applicator and to the consumer of treated products; and effects on non-target species including those of aquatic ecosystems.

Guidelines for research into effects of pesticides on aquatic life are of concern to this Panel. In view of documented effects of pesticides on aquatic life and the apparently ubiquitous distribution of certain pesticides in fish (Johnson 1968,[158] Henderson, Johnson and Inglis 1969,[155] Mollison 1970[172]), it seems reasonable to conclude that existing guidelines are not sufficient. Mount (1967)[173] reported that there were numerous studies on toxicological and physiological effects of pesticides in fish, but that the data were inadequate because of several common deficiencies. Further, he concluded that there was a paucity of data that could be used to correlate toxicological, physiological, or analytical findings with significant damage to aquatic forms. Therefore, research guidelines for predicting potential hazards of pesticides to be used in, or those with a high probability for contamination of aquatic communities must result in findings that are relatable within the scientific disciplines concerned.

Guidelines for research and objectives suggested by this Panel are:

(1) to provide a research framework that generates anticipatory rather than documentary information concerning effects of pesticides on aquatic communities;

(2) to encourage research that is directly applicable to the process of pesticide registration.

The framework (Figure II-E-1) is designed with fish a the primary test animal(s). However, it is also compatible with parallel investigations intended to provide data essential to the protection of fish-food organisms. In all cases, sufficient numbers of individuals and replications must be included to estimate statistical significance of results. All studies should report sources, physical quality, disease treatments, and holding conditions (photoperiod, diet and feeding rate, water quality) of test animals. The Panel recommends that chemical analyses be performed on test animals, diets, and holding waters to document pre-exposure of test animals to pesticides or other contaminants. Analytical methods should include results for reagent blanks, and they should document limits of sensitivity, detection, reproducibility, and recovery efficiency for extracts.

The guidelines are general and are not intended to limit research nor to present specific methods. If pesticide investigations can be tailored, at least in part, along accepted guidelines, then a much greater reservoir of interrelated anticipatory data will become available for the purpose of registering pesticides and establishing water quality criteria. All, or parts of the guidelines, may be utilized by an investigator depending upon: the capacity of his laboratory and staff; extent and applicability of biological or chemical data already available; intended use pattern(s) and target(s) of the pesticides; or research objectives other than registration.

I. PRINCIPAL SYSTEMS

A. Acute Toxicity: Static Bioassay (Litchfield and Wilcoxon 1949,[164] Lennon & Walker 1964,[163] Nebeker & Gaufin 1964,[176] Sanders and Cope 1966,[180] Burdick 1967,[144] Sprague 1969,[182] Schoettger 1970,[181] Environmental Protection Agency 1971).[150]

1. Purpose

The limitations of static bioassays are recognized; however, they do provide the first, and probably quickest, index of relative toxicity. Further, they are useful in estimating the relative influence of variables such as species susceptibility, temperature, pH, water quality, and rate of chemical deactivation on toxicity. Thus, acute static bio-

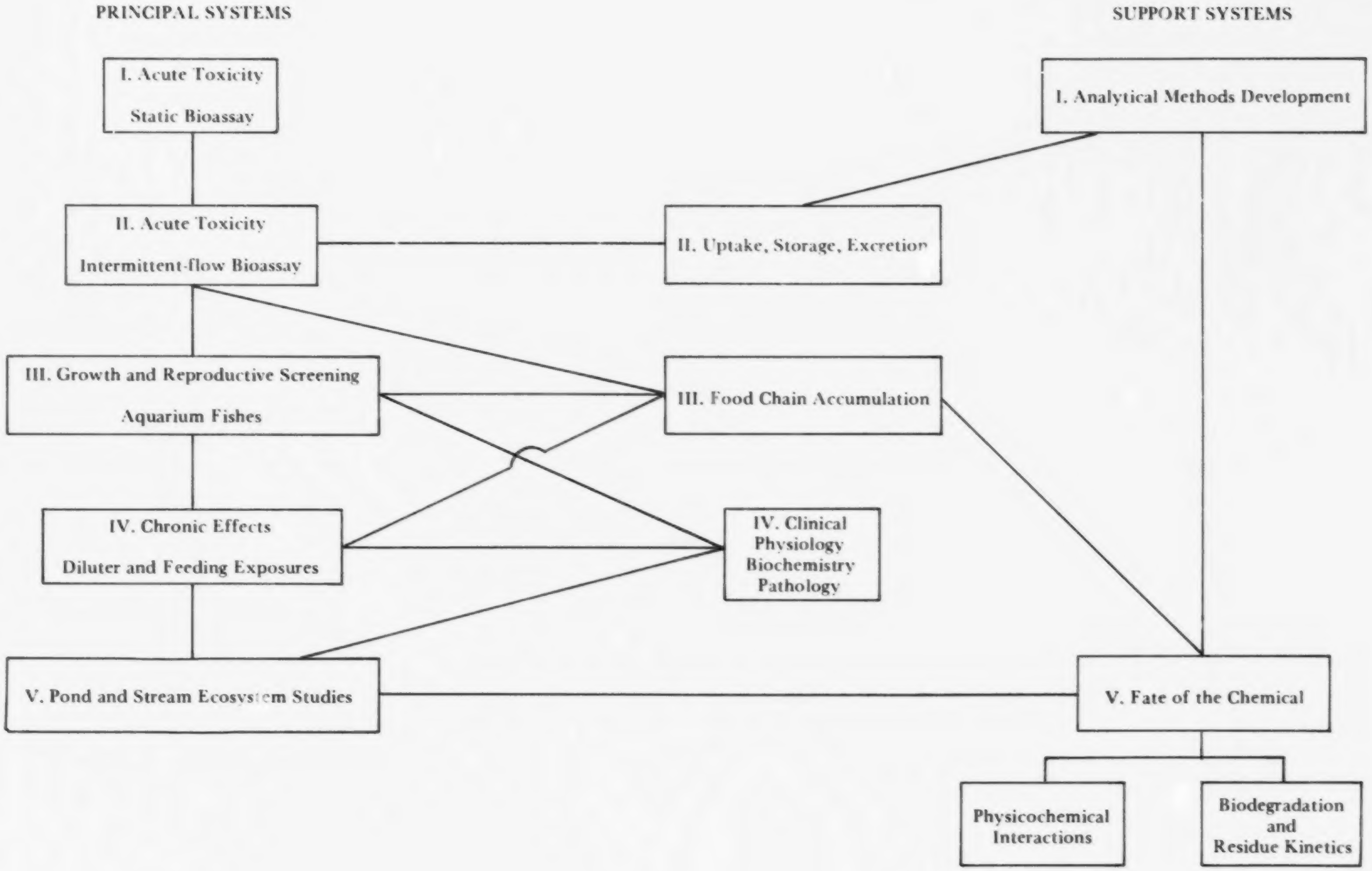
PRINCIPAL SYSTEMS
SUPPORT SYSTEMS
I. Acute Toxicity Static Bioassay
II. Acute Toxicity Intermittent-flow Bioassay
III. Growth and Reproductive Screening Aquarium Fishes
IV. Chronic Effects Diluter and Feeding Exposures
V. Pond and Stream Ecosystem Studies
I. Analytical Methods Development
II. Uptake, Storage, Excretion
III. Food Chain Accumulation
IV. Clinical Physiology Biochemistry Pathology
V. Fate of the Chemical
Physicochemical Interactions
Biodegradation and Residue Kinetics

FIGURE II-E-1—A Research Framework[a]

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[a] See text for discussion.

assays are essential to delineate prerequisites for chronic studies.

2. Scope

a. Initial bioassay

These studies are conducted with technical and formulated pesticides using one type of water (reconstituted). The 96-hour LC50 (tolerance limit for 50 per cent of the test animals) is determined for rainbow trout (*Salmo gairdneri*) at 12 C, and for bluegills (*Lepomis machrochirus*), fathead minnows (*Pimephales promelas*), and channel catfish (*Ictalurus punctatus*) at 22 C. Suggested species of invertebrates included daphnids (*Daphnia magna*), glass shrimp (*Palaemonetes kadiakensis*), scud (*Gammarus pseudolimnaeus*), and midge larvae (*Chironomus plumosus*).

b. Definitive bioassay

Bioassays conducted as described above. Trout are tested at 7 C and 17 C, whereas bluegills, fathead minnows, and channel catfish are tested at 17 C and 27 C. Water quality (reconstituted) is modified to include soft and hard waters, and water of *ca.* pH 6 and 9 (Marking and Hogan 1967,[167] Berger, Lennon and Hogan 1969).[139] Other temperatures and potentially threatened species must be added or substituted depending upon specific conditions under which the pesticide is to be used.

c. Deactivation index

Several series of test concentrations as in a or b are prepared and stored for appropriate intervals, such as 1, 2, 4, 8, 16 N days. After storage, the solutions are bioassay conducted at the same time. Division of the reference 96-hour LC50 values by the values for stored solutions gives an estimate of rate of pesticide deactivation when plotted against storage time. Additional trials may be required to determine effects of variables such as pH, temperature, light. Residue analyses of stored solutions provide excellent support data for measures of biological deactivation.

B. Acute Toxicity: Intermittent-flow Bioassay Jensen & Gaufin 1964,[157] Mount and Brungs 1967,[174] (Standard Methods 1971).[185]

1. Purpose

Intermittent-flow bioassays are designed to minimize or overcome deficiencies characteristic of static bioassays, and are particularly suited for long exposures of test animals to pesticides with low water solubilities. Specialized apparatus is required for such studies, but results are generally considered more reliable, and more representative of actual toxicity than those derived from static bioassay. Nevertheless, the speed and flexibility of the latter make them essential in establishing operational designs (e.g., water quality, temperature, species) for the former method.

2. Scope

a. 96-hour LC50

This is a standard bioassay and is obtained with any water supply (analyzed for chemical characteristics) suitable to the selected test species. When variables such as temperature or water quality affect toxicity (as determined in sections IA2b and IA2c), flowing bioassays must be designed accordingly. In some instances, a design consistent with water quality and species in the locality of pesticide use may be appropriate. Because intermittent-flow bioassays require analyzed concentrations (rather than calculated values), analytical methods must be developed prior to the start of bioassays. The use of radio-labeled pesticides greatly assists analysis. Also, test animals treated with radioactive pesticides are invaluable for preliminary estimates of pesticide uptake, storage, and excretion. In addition, gross observations should be made for pathological and behavioral changes.

b. Lethal threshold concentration (Threshold LC50)

The Threshold LC50 is estimated subsequent to determination of the 96-hour LC50 and may require lower concentrations. In general, the bioassay is conducted as in IB2a, but contined in 48-hour increments after the 96-hour observation period. The Threshold LC50 is determined when further mortality has ceased in all test tanks, compared to the control. If toxicant-related mortality continues beyond 30 to 60 days, the bioassay may be discontinued and the LC50 reported according to the test duration. Pesticide uptake, storage, and excretion studies may be more meaningful, when conducted on test animals, from these studies than on those exposed for only 96 hours.

C. Growth and Reproductive Screening: Aquarium Fishes (Hisaoka and Firlit 1962,[156] Clark and Clark 1964,[145] Breeder and Rosen 1966[142]).

1. Purpose

Mount (1967)[173] indicated that growth and reproduction of fish were important in assessing safe concentrations of pesticides, and could be determined within one year. However, when estimates of potential hazards are needed for a relatively large number of pesticides, and space and time are limited, tests using fish with short life cycles may be desirable for establishing priorities for later research. Species such as the ovoviviparous guppy (*Poecilia reticulata*) and oviparous zebrafish (*Brachydanio rerio*) produce numerous progeny that may reach sexual maturity within six weeks under laboratory conditions. Thus, effects of pesticides may be followed through several generations within a short time.

2. Scope

Zebrafish and guppies are exposed to pesticides in intermittent-flow diluters. Also, the pesticide may be incorporated into their diets if food chain studies suggest that dietary uptake is a potentially significant route of exposure. Observations are made on mortality, growth, egg production, and hatchability, and on incidence of offspring anomalies (e.g., terata, mutations).

D. Chronic Effects: Diluter and Feeding exposures (Burdick, et al., 1964,[145] Macek 1968,[165] Eaton 1970,[148] Environmental Protection Agency 1971,[150] Johnson et al. 1971[159]).

1. Purpose

In general, these studies are conducted as in IC2 and are central to predicting safe concentrations of pesticides to sport, commercial, or forage fishes, and to fish-food organisms.

2. Scope

Chronic studies may either include the complete life cycle or a portion of the cycle. Full chronic studies are conducted currently with fathead minnows, daphnids, and scuds and involve continuous exposures of eggs, juveniles and adults. Rainbow trout, brook trout (*Salvelinus fontinalis*), channel catfish, bluegills, and largemouth bass (*Micropterus salmonides*) are used in partial chronics, and adults are exposed continuously through spawning. Flow-through bioassays are performed by exposing the test animals to pesticides (or degradation products) in water, in their diets, or both, depending upon relative stability of the pesticide and its tendency to accumulate in fish-food organisms. Where profiles of pesticide degradation in water are established, studies simulating degradation should be incorporated into the concentration spectra by periodic modification of toxicant solutions (concentration and composition). Exposures should include the reproductive phase or a selected interval prior to reproduction depending upon species and anticipated time of pesticide application. Chronic studies should evaluate effects on growth, and on natural and artificial reproduction. Studies with invertebrates should include measured effects on metamorphosis and reproduction. Clinical observations on physiological, biochemical, and pathological effects, as well as analyses for residues, degradation products, and residue kinetics should be correlated with effects on growth and reproduction.

E. Pond and Stream Ecosystem Studies (Cope, et al. 1970,[147] Kennedy et al. 1970,[161] Kennedy and Walsh 1970,[160] Lennon and Berger 1970[162]).

1. Purpose

Laboratory estimates of safe pesticide applications must be confirmed by controlled research in lentic and lotic ecosystems. Therefore, ponds or artificial streams are invaluable in studying the impact of pesticides under interacting physical, chemical, and biological conditions.

2. Scope

Applications of pesticides are made according to anticipated rate and use patterns. However, concentration spectra should include both excessive rates, and rates estimated to be safe in laboratory studies. Species used in the studies should approximate those found in intended areas of pesticide usage. Factors to be studied include:

a. mortality
b. growth
c. reproductive success
d. gross behavior
e. clinical physiology, biochemistry and pathology
f. invertebrate metamorphosis
g. species diversity
h. trophic level production
i. energy transfer
j. fate of the chemical

II. SUPPORT SYSTEMS

A. Chemical Methods Development

1. Purpose

Residue analyses of water and of fish and fish-food organisms exposed to pesticides are potent indicators of probable biological accumulation or degradation of these chemicals. Biological systems used in the primary research framework easily provide study materials which permit correlations between biological effects and residues. The use of radio-labeled pesticides early in the research framework quickly pinpointed location of the pesticide and degradation products and greatly assisted refinement of analytical methods. Various combinations of isolation and identification techniques are required to analyze metabolites or degradation products in test animals exposed chronically to pesticides.

2. Scope

Methods may begin with acute, static bioassays for deactivation indexes (IA2c) or later with acute, intermittent-flow bioassays. The studies are expanded as dictated by interpretation of results. Concentrations of ^{14}C-, ^{36}Cl-, ^{32}P-, or ^{35}S-labeled pesticides are determined radiometrically without extraction and cleanup (Hansen and Bush 1967,[154] Nuclear-Chicago Corporation 1967,[177] Biros 1970a[140]). At least four test animals (including fish) should be collected at five intervals during the pesticide exposure to estimate uptake and degradation rates. For smaller organisms, a minimum of 100 milligrams of wet sample are required. After development and refinement of analytical methods, spot checks of radioactive samples will confirm residues. Analyses of metabolites and degradation products require that sample extracts be cleaned up with gel permeation or adsorption chromatography (U. S. Department of Health Education and Welfare 1968,[187] 1969,[188] Stalling, Tindle and Johnson 1971,[184] Tindle 1971).[186] Radioactive residues must be characterized by TLC autoradiography, and further identified by gas chromatography-mass spectrometry (GC-MS) or other spectroscopic methods (Biros 1970b,[141] Stalling 1971).[183]

B. Uptake, Storage and Excretion

1. Purpose

Investigations of chemical residues are undertaken early in the research framework to obtain a working perspective of pesticide persistence, degradation, and bioconcentration in aquatic organisms. The studies should attempt to correlate residue kinetics with toxicology and chronic effects. Thus, later research can be better designed to assess interactions of pesticides with fish, fish-food organisms, and water quality (ID).

2. Scope

The studies should include:

a. radiometric or chemical analyses or both, of test animals at intervals during acute, intermittent-flow bioassays to determine rates of accumulation and residue plateaus;

b. determination of biological half-life of accumulated residues after termination of exposure (Macek et al. 1970);[166]

c. determination of degree of pesticide degradation in water and test animals by comparing residues of radioactive materials with concentration of parent chemical, measured chemically (Johnson et al. 1971,[159] Rodgers and Stalling 1971[179]). (Autoradiograms of thin-layer chromatographic plates may provide the initial data on degradation products.)

C. Food-Chain Accumulation (Brock 1966,[143] Johnson et al. 1971,[159] Metcalf et al. 1971).[170]

1. Purpose

The functions of laboratory food chain studies include: estimates of propensity for pesticide (or its degradation product), uptake by each member of a 3-component food chain, estimates of potential pesticide transfer to higher trophic levels, and determinations of residue concentrations likely to be encountered in forage of fish. (Residue values are used in formulating pesticide-containing diets, section IC and ID.)

2. Scope

A suggested laboratory food chain may be composed of: an appropriate primary producer (green algae) such as *Scenedesmus*, *Ankistrodesmus* and *Chlorella* Spp.: or decomposers (bacteria) such as *Aerobacter*, *Bacillus*, *Achromobacter*, *Flavobacter*, *Aeromonas* Spp.; a primary consumer such as *Daphnia magna*, *D. pulex*, or other suitable microcrustacea; and a secondary consumer such as fathead minnows or small bluegills, largemouth bass, rainbow trout. Members of the food chain are exposed to radio-labeled pesticides in diluters (or other constant-flow devices) at concentrations appropriate for the most sensitive element. Rate of uptake and residue plateau are measured radiometrically and the identities of parent compound or degradation products are confirmed by chemical methods, whenever possible. The potentials for biotransfer and biomagnification are determined by feeding pesticide-treated lower members to higher trophic levels with and without concurrent water exposures. An alternative, but less desirable, type of feeding trial would utilize artificial foods fortified with appropriate amounts of pesticide.

D. Clinical: Physiology, Biochemistry, Pathology (Mattingly 1962,[169] Mattenheimer 1966,[168] Natelson 1968,[175] Pickford and Grant 1968,[178] Grant and Mehrle 1970,[153] Mehrle 1970[171]).

1. Purpose

Clinical studies are most closely associated with chronic investigations of pesticidal effects on growth and reproduction. It is likely that these effects are expressions of earlier, more subtle physiological, biochemical, or pathological dysfunctions. Thus, selected clinical examinations may reveal correlations that are useful in early detection of adverse effects. These studies may also reveal impaired homeostasis mechanisms for compensating ephemeral environmental stresses (e.g., oxygen deficiency, starvation, exercise, rapid changes in temperature, pH, salinity) that are not otherwise anticipated in this reserach framework.

2. Scope

Routine clinical studies are impractical during full chronic investigations with fathead minnows (and other small test animals), because of their small size and the difficulty in collection of adequate amounts of tissue. However, at hatching, young are observed for incidence of abnormalcy; and other young removed for thinning, should be used in histocytological examinations and stress tests. The latter tests measure relative survival under stresses such as those mentioned in IID1 above. Individuals from partial chronic and pond or stream studies are also examined and tested as in full chronic studies. Because of larger size, they are useful in clinical studies. These studies, however, are not necessarily intended as ends in themselves. Examples of appropriate clinical examinations include:

a. stress response—induced production of cortisol by injection of adrenocorticotrophic hormone (purified mammalian ACTH);

b. thyroid activity—^{125}iodine (^{125}I) uptake;

c. osmoregulatory ability—serum sodium, chloride, and osmolality;

d. diagnostic enzymology—clinical analyses for activities of liver and serum glutamate-oxaloacetate transaminase, glutamate-pyruvate transaminase, glutamate dehydrogenase, alkaline phosphatase, and lactate dehydrogenase;

e. ammonia detoxifying mechanism (brain and liver glutamate dehydrogenase, brain glutamine synthetase, and ammonia concentrations in brain and serum);

f. cholinesterase activity of serum and brain;

g. general nutritional state and activity of microsomal and mitochondrial enzymes—injection of ^{14}carbon-labeled glucose and relative evolution of $^{14}CO_2$ by liver; and

h. histocytological examinations of liver, brain, pancreas, gill, and kidney by light and electron microscopy.

E. Fate of the Chemical

1. Purpose

The environmental fate of a pesticide is determined by its interactions with physicochemical and biological processes. Its distribution is the result of partition between the biota and sedimentation processes, and degradation rates associated with each of these. Segmentally, these studies attempt to predict the relative ecodistribution of pesticides, identify physicochemical and biological degradation products, and describe their kinetics. Biological effects of these compounds must be correlated with residues in order to anticipate their ecological impact under the conditions of use.

2. Scope

a. Biodegradation and Residue Kinetics

Fish and invertebrates—these studies on residue degradation and uptake are more definitive than the initial uptake studies involved in acute intermittent-flow bioassays. Equilibrium of the residues (parent compound or metabolites or both) in the organisms during the exposure period must be documented to strengthen correlation of exposure concentrations and biological effects. Special consideration must be given to multiple component pesticides. Both the composition and isomer ratios can be altered and should be included in determining safe levels of pesticide exposure. The chemical burden and kinetics of uptake in the test organism are determined by sampling at not less than four intervals during the test exposures. No less than three fish or other samples per concentration are analyzed at each sampling period.

Gas-liquid chromatography (GLC) and Gas-liquid chromatography-mass spectrograph (GLC-MS) analyses are then made on each sample to determine which fractions of the radioactive residues are attributed to the parent compound(s) and what changes occurred in the composition and isomer-ratios of the pesticide. Thin layer chromatographic examination of nonvolatile metabolites is recommended for compounds which cannot be analyzed by GLC (Biros 1970b,[141] Johnson et al. 1971[159]).

Chemical information obtained from the various invertebrate organisms is examined in light of possible impact on the food chain of fish and other organisms. These data give an estimate of the relative importance of bioconcentration, biopassage, and biodegradation in the various trophic levels in predicting the effect on ecosystems (Eberhardt, Meeks, and Peterle 1971).[149]

Microorganisms—These studies are designed to ascertain whether or not a pesticide or its degradation product(s) is biodegradable by microorganisms in an aquatic environment (Faculty of American Bacteriologists 1957).[151] Benthic muds are incubated with the pesticide (or degradation product(s) or both) in liquid culture. One sample is sterilized to distinguish chemical or biological degradation, or both. Variables investigated concerning the basic microorganism-pesticide interaction during incubation are:

- duration: 1-3-7-14-21-30 days;
- temperature: 15-25-35 C;
- pH: 5.0-7.0-9.0;
- oxygen tension: aerobic or anaerobic (nitrogen overlay).

b. Physicochemical Interactions

These studies are designed to determine the interactions of water quality factors as they affect rates of sorption, desorption, and loss of chemicals from the aquatic system, and chemical modifications of the parent compound. These data permit accurate assessment of the biological availability to, and effects of the subject chemical on, the aquatic biota.

Sediment binding studies (i.e., sorption, desorption rates) should consider the effects of as many combinations of the following as possible:

- pH: 6, 7.5, 9;
- hardness: 10, 45, 300 ppm as $CaCO_3$;
- temperature: 7, 17, 27 C;
- sediment type (heavy, light, high/low-organic); binding profile, i.e., degree of binding as a function of particle size and composition.

Chemical degradation rates as influenced by the previous characteristics should also be analyzed. In addition, the importance of photodegradation (visible and ultraviolet) must also be examined. Product identification will utilize analyses by GLC, mass spectrometry, and infrared spectrometry. Degradation products will be synthesized where necessary for biological or chemical testing. Volatization and loss of pesticides from the aqueous system must also be considered, particularly where factors of pH or temperature are important.

APPENDIX II-F

Pesticides Recommended for Monitoring in the Environment[f]

Common or trade name	Chemical name[a]
Aldrin	not less than 95 percent of 1,2,3,4,10,10-hexachloro-1,4,4a,5,8,8a-hexahydro-1,4-endo-exo-5,8-dimethanonaphthalene
Amitrole	3-amino-s-triazole
Arsenic-containing pesticides (Inorganic and organic)	
Atrazine	2-chloro-4-(ethylamino)-6-(isopropylamino)-s-triazine
Azinphosmethyl (Guthion®)	O,O-dimethyl phosphorodithioate S-ester with 3-(mercaptomethyl)-1,2,3-benzotriazin-4(3H)-one
Benzene hexachloride (BHC)	1,2,3,4,5,6-hexachlorocyclohexane, consisting of several isomers and containing a specified percentage of gamma isomer[b]
Captan	N-[(trichloromethyl)thio]-4-cyclohexene-1,2-dicarboximide
Chlordane	at least 60 percent of 1,2,4,5,6,7,8,8-octachloro-3a,4,7,7a-tetrahydro-4,7-methanoindan and not over 40 percent of related compounds
2,4-D (including salts, esters, and other derivatives)	(2,4-dichlorophenoxy)acetic acid
DDT (including its isomers and dehydrochlorination products)	1,1,1-trichloro-2,2-bis(p-chlorophenyl)ethane; technical DDT consists of a mixture of the p,p′-isomer and the o,p′-isomer (in a ratio of about 3 or 4 to 1)
Dicamba	3,6-dichloro-o-anisic acid
Dieldrin	not less than 85 percent of 1,2,3,4,10,10-hexachloro-6,7-epoxy-1,4,4a,5,6,7,8,8a-octahydro-1,4-endo-exo-5,8-dimethanonaphthalene
Dithiocarbamate pesticides:	
Maneb	[ethylenebis(dithiocarbamato)]manganese;
Ferbam	tris(dimethyldithiocarbamato)iron;
Zineb	[ethylenebis(dithiocarbamato)]zinc;
Endrin	1,2,3,4,10,10-hexachloro-6,7-epoxy-1,4,4a,5,6,7,8,8a-octahydro-1,4-endo-endo-5,8-dimethanonaphthalene
Heptachlor	1,4,5,6,7,8,8-heptachloro-3a,4,7,7a-tetrahydro-4,7-methanoindene
Heptachlor epoxide	1,4,5,6,7,8,8-heptachloro-2,3-epoxy-3a,4,7,7a-tetrahydro-4,7-methanoindan
Lindane	1,2,3,4,5,6-hexachlorocyclohexane, gamma isomer of not less than 99 percent purity
Malathion	diethyl mercaptosuccinate S-ester with O,O-dimethyl phosphorodithioate
Mercury-containing pesticides (Inorganic and organic)	
Methoxychlor	1,1,1-trichloro-2,2-bis(p-methoxyphenyl)ethane; technical methoxychlor contains some o,p′-isomer also
Methyl parathion	O,O-dimethyl O-(p-nitrophenyl) phosphorothioate
Mirex	dodecachlorooctahydro-1,3,4-metheno-1H-cyclobuta[cd]pentalene
Nitralin (Planavin®)	4-(methylsulfonyl)-2,6-dinitro-N,N-diproylaniline
Parathion	O,O-diethyl O-(p-nitrophenyl) phosphorothioate
PCNB	pentachloronitrobenzene
Picloram	4-amino-3,5,6-trichloropicolinic acid
Silvex (including salts, esters, and other derivatives)	2-(2,4,5-trichlorophenoxy)propionic acid
Strobane®	terpene polychlorinates containing 65 percent chlorine
2,4,5-T (including salts, esters, and other derivatives)	(2,4,5-trichlorophenoxy)acetic acid
TDE (DDD) (including its isomers and dehydrochlorination products)	1,1-dichloro-2,2-bis(p-chlorophenyl)ethane; technical TDE contains some o,p′-isomer also
Toxaphene	chlorinated camphene containing 67–69 percent chlorine
Trifluralin	α,α,α-trifluoro-2,6-dinitro-N,N-dipropyl-p-toluidine
Secondary List of Chemicals for Monitoring	
DCNA (Botran®)	2,6-dichloro-4-nitroaniline
Carbaryl	1-naphthyl methylcarbamate
Demetron (Systox®)	mixture of O,O-diethyl S (and O)-[2-(ethylthio)ethyl] phosphorothioates
Diazinon	O,O-diethyl O-(2-isopropyl-6-methyl-4-pyrimidinyl) phosphorothioate
Disulfoton (Di-Syston®)	O,O-diethyl S-[2-ethylthio)ethyl]phosphorodithioate
Diuron	3-(3,4-dichlorophenyl)-1,1-dimethylurea
Endosulfan (Thiodan®)	1,4,5,6,7,7-hexachloro-5-norbornene-2,3-dimethanol cyclic sulfite
Fenac[c]	(2,3,6-trichlorophenyl)acetic acid
Fluometuron	1,1-dimethyl-3-(α, α, α-trifluoro-m-tolyl)urea
Inorganic bromide from bromine-containing pesticides	
Lead-containing pesticides such as lead arsenate	
Linuron	3-(3,4-dichlorophenyl)-1-methoxy-1-methylurea
PCP	pentachlorophenol
Propanil[c]	3′,4′-dichloropropionanilide
Triazine-type herbicides[d]:	
Simazine	2-chloro-4,6-bis(ethylamino)-s-triazine;
Propazine	2-chloro-4,6-bis(isopropylamino)-s-triazine;
Prometryne	2,4-bis(isopropylamino)-6-(methylthio)-s-triazine
TBA	2,3,5-trichlorobenzoic acid, usually available as mixed isomers
List of Special Chemicals for Monitoring[e]	
Polychlorobiphenyls (PCBs)	Mixtures of chlorinated biphenyl compounds having various percentages of chlorination.
Polychlorodibenzo-p-dioxins	Dibenzo-p-dioxins having various degrees of chlorination such as the tetra-, hexa-, or octachlorodibenzo-p-dioxins, present as impurities in various chlorine-containing phenols and early samples of 2,4,5-T.

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[a] Chemical names are in accordance with Chemical Abstracts.

[b] Report individual isomers when possible.

[c] Some compounds are used primarily on one or two crops or in certain regions rather than country-wide; for example, the herbicides fenac and propanil are used mainly on sugar cane and rice, respectively.

[d] Note that atrazine has been moved to the Primary List.

[e] This list contains chemicals which, although not considered to be pesticides themselves, are of special interest in monitoring studies.

[f] Schechter, 1971.[183]

APPENDIX II-G

TOXICANTS IN FISHERY MANAGEMENT

There is much evidence that primitive people in Asia and South America used poisonous plants to capture freshwater and saltwater fishes for food. In China, extracts from toxic plants have been employed for thousands of years to remove undesirable fish from ponds under intensive fish culture. The practice of applying toxicants in sport fishery management of waters by poisoning non-game fish has been used as a management tool (Prévost 1960).[193] Some of the many causes and instances of fishes in pest situations were discussed by Lennon (1970).[191]

A survey commissioned by the Food and Agriculture Organization of the United Nations in 1970 disclosed that 29 countries on the five continents are using toxicants in the culture or management of food and game fishes (Lennon et al. 1970).[192] Forty-nine of the 50 states in the United States and most provinces in Canada have used or are using piscicides in fishery programs. The toxicants are employed to correct various problems in farm, ranch, and fish-production ponds; in natural lakes and reservoirs; and in streams and rivers.

The chemicals that served most commonly as fish toxicants since the 1930's were basically insecticides in nature and formulation Rotenone and toxaphene, for example, were applied predominantly as piscicides in the United States and Canada in 1966 (Stroud and Martin 1968),[194] but several dozens of chemicals including natural poisons, inorganics, chlorinated hydrocarbons, and organophosphates have had testing or use to kill fish (Lennon et al. 1970).[192]

There is a significant change in the use of toxicants in fishery management. Increasing concerns by the public and government regarding broad spectrum, persistent pesticides have resulted in stiff requirements for registration of fish toxicants and regulation of their use in public waters. Well justified emphasis is being placed now on the development and use of piscicides that are specific to fish, harmless at use levels to non-target plants and animals, non-persistent in the aquatic environment, and safe to handle and apply. An enormous amount of research is required now to secure or retain registration of a fish toxicant. The research includes long-term studies on safety to man and mammals, on efficacy to target fish, on residues in fish and other aquatic life, and on degradation or deactivation of the toxicant in the environment.

Programs for the management of public waters are being more closely scrutinized for any temporary or long-term effects they will have on the environment. More emphasis is being placed on the enhancement and protection of the integrity of ecosystems as the main goal for management of our living resources. The importance of preserving a diversity of aquatic habitats and natural communities as important gene pools, which may be of inestimable value to mankind in the future, as well as for education, research, and aesthetic enjoyment must be clearly recognized. If control measures are undertaken which will kill non-target aquatic species (fish or invertebrates), then careful consideration should be given to preserving populations of these species for restrocking in order to reestablish stability of the community. Furthermore, more attention should be given to beneficial use of nuisance populations of aquatic organisms and efficient harvesting methods should be developed as part of any integrated control program.

There are five divisions of the management process that must be considered by fishery managers and project review boards. They are:

Demonstration of need

A fishery problem is at first presumed to exist, then studied and defined, and proven or disproven. If proven, the need for immediate or eventual correction is assessed and weighed against all possible environmental, scientific, and political considerations. The need then is documented and demonstrated to those in a decision-making capacity.

Selection of method(s) for solution of problem

All possible solutions to the problem by means of chemical, biological, physical, and integrated approaches must be considered and evaluated in terms of effectiveness on target fishes, safety to non-target plants and animals, and environmental impact. An important rule of thumb is that a toxicant should be used only as a last resort.

The selection of an approach to solve the problem, therefore, must be accomplished on the basis of sound fact-finding and judgment. Every opportunity for exploiting an integrated approach to management and control deserves consideration to protect the integrity of ecosystems.

The selection of an approach to management of native fish populations and control of exotic species should be approved by an impartial board of review.

Selection of a toxicant

If a chemical approach to solution of the problem is chosen, the next major step is selection of the correct toxicant. The toxicant must be one registered for the use, specific to the target species, and relatively compatible with the environmental situation.

Method of Application

The proximity of application transects on lakes or metering stations on streams is an important consideration. Application points must be close enough together to avoid locally excessive concentrations that may be harmful to non-target life.

Every opportunity to achieve *selective* action on target organisms by adjusting the application method or timing should be exploited.

Pre- and post-treatment assessments

Careful surveys and assessments of the target and non-target life in the problem area are needed prior to a treatment. The data must be quantitatively and qualitatively representative.

The actual application must be preceded by competent ecosystem study of the habitat to be treated. Moreover, on-site bioassays of the candidate toxicant must be conducted against representative target and non-target organisms collected in the problem area. The dose (concentration plus duration of exposure) of toxicant needed for the reclamation is calculated from the results of the on-site bioassays.

Following an application, thorough ecosystem studies and assessments of target and non-target life must be made in the problem area. Some surveys should be accomplished immediately; others should be prosecuted periodically for 1 to 2 years to evaluate the effect of the treatment to determine if the original problem was corrected, and to detect any long-term and or adverse effects on non-target life and the environment in general.

All chemical treatments of public waters should be reviewed by impartial boards at appropriate state and federal levels. Resource administrators, managers and scientists in fisheries, wildlife, ecology, and recreation should be represented on the boards, and they should call in advisors from the private and public sectors as necessary to evaluate proposed projects realistically and fairly. A board must have decision-making authority at each step of the treatment process; thus, a smoothly working system for getting facts from the field to the board and its decisions back to the field is necessary. Furthermore, a review board must have continuity so that it can assess the results of preceding treatments and apply the experience obtained to subsequent management activities.

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PREFACE

Tables 1-3 in this Appendix have been compiled to provide information on the effects of inorganic constituents on marine organisms. Data on bioassay tests with fresh water organisms are included, especially when the information concerning marine organisms is inadequate. This was also done when the same investigator studied both fresh water and marine organisms. The substances tested are listed in alphabetical order, generally based upon the constituent in the compound considered to be critical. The entries are arranged within substances by year of publication and author. The units used are those presented in the original publication. In some cases it is impossible to know whether the concentration is expressed in terms of the element or the compound tested, but if the information was presented in the original publication, it is so indicated. The organism used in the test is identified as in the original reference, giving the specific name wherever it is available. Very abbreviated descriptions of the conditions of the test are presented. The value of the compilation is to indicate the range of concentrations tested, the species used, and the references to the original work. The reader is urged to refer to the original reference for more precise details about the test conditions or to the author if the necessary details were omitted in the publication.

Generally, in Table 1 the acute dose for a 96 hr LC50 is presented. If the time of the test was different, it is indicated in parentheses after the concentration listed, for example, (48 hr).

L = Laboratory bioassay
  BS = bioassay static
  BCF = bioassay continuous flow
  BA = bioassay acute
  BCH = bioassay chronic
a = water temperature
b = ambient air temperature
c = pH
d = alkalinity (total, phenolphthalein or caustic)
e = dissolved oxygen
f = hardness (total, carbonate, Mg or CaO)
g = turbidity
h = oxidation reduction potential
i = chloride as Cl
j = BOD, 5 day; (J) = BOD, short-term
k = COD
l = Nitrogen (as NO_2 or NO_3)
m = ammonia nitrogen as NH_3
n = phosphate (total, ortho-, or poly)
o = solids (total, fixed, volatile, or suspended)
p = CO_2
  BOD = biochemical oxygen demand

TABLE 1—Acute dose of inorganic chemicals for aquatic organisms

Constituent	Acute dose 96 hr LC50	Species	Conditions	Literature citation*
Aluminum (Al)	250 ppm	Micropterus salmoides	$Al(SO_4)_3$; 18 H_2O; pH 7.2-7.6; 64-8 ppm	Sanborn 1945[108]
		fish and river crab		Podubsky and Stedronsky 1948[96]
	88 ppm (few Hrs)	Sebastes marinus		Pulley 1950[100]
	17.8 mg/l (short time)	Sebastes marinus	$AlCl_3$; sea water	Pulley 1950[100]
	235 mg/l	Gambusia affinis	19–22 C; turbid water; turbidity 235 to 25 mg/l; $Al_2(SO_4)_3 \cdot 18\,H_2O$	Wallen et al. 1957[133]
	133 mg/l	Gambusia affinis	highly turbid water	Wallen et al. 1957[133]
	240 ppm (48 hr)	Gambusia affinis	$AlCl_3$, static acute bioassay turbid water; a,c,d,e,g	Wallen et al. 1957[133]
	135 mg/l (48 hr)	Gambusia affinis	$AlCl_3$, static acute bioassay turbid water; a,c,d,e,g	Wallen et al. 1957[133]
Ammonia (NH_3)	18.5 mg/l (48 hrs)	Lepomis macrochirus	tap water; reoxygenated 20 C; NH_4OH	Turnbull et al. 1954[130]
	15 mg/l (48 hr)	″	conc. as NH_4OH; tap water; 20 C.	″
	6.0 ppm	Lepomis macrochirus	continuous flow, acute bioassay, a,c,e,f; aerated distilled NH_4Cl	Cairns Jr. and Scheier unpublished 1955[142]
	300 mg/l (6 hrs)	minnows	hard water; NH_4Cl	LeClerc and Devaminck 1955[73]
	4000–5000 mg/l (6 hrs)	minnows	distilled water; NH_4Cl	LeClerc and Devaminck 1955[73]
	8.0 mg/l (time not specified)	Daphnia		Meinck et al. 1956[79]
	17.5 mg/l (48 hr)	Pimephales promelas	conc. as NH_4OH; tap water;	Black et al. 1957[13]
	7.7 ppm	Lepomis macrochirus	NH_4Cl; distilled aerated water; static acute bioassay; a,c,d,f. NH_4Cl as N;	Cairns Jr. and Scheier 1957[25]
	248 mg/l	Gambusia affinis	21 C; in turbid water using $(NH_4)_2S$	Wallen et al. 1957[133]
	490 mg/l	Gambusia affinis	in turbid water; NH_4Cl	Wallen et al. 1957[133]
	240 mg/l	″	using $(NH_4)_2SO_3 \cdot H_2O$: 20–21 C, turbidity lowered from 220 to 25 mg/l	″
	114 mg/l	Gambusia affinis	using NH_4SCN; turbid water 16–23 C	Wallen et al. 1957[133]
	1290 mg/l	Gambusia affinis	turbid water; 20–21 C using $(NH_4)_2SO_4$; reduced turbidity from 240–25 mg/l	Wallen et al. 1957[133]
	240 mg/l	Gambusia affinis	highly turbid water; $(NH_4)_2CrO_4$	Wallen et al. 1957[133]
	136 mg/l	″	″ $(NH_4)_2Cr_2O_7$	″
	37 mg/l	Gambusia affinis	turbid water	Wallen et al. 1957[133]
	910 mg/l (24 hr)	Gambusia affinis	using NH_4SCN; turbid water 16–23 C	Wallen et al. 1957[133]
	238 ppm (2 day)	Gambusia affinis	static acute bioassay, a,c,d,e,g; ammonium acetate; high turbidity pH 7.6–8.8.	Wallen et al. 1957[133]
	238 ppm (2 day)	Gambusia affinis	same as above using $(NH)_4CO_3$	″
	510 ppm (2 day)	Gambusia affinis	static acute bioassay, a,c,d,e,g, high turbidity; NH_4Cl	Wallen et al. 1957[133]
	270 ppm (2 day)	Gambusia affinis	static acute bioassay, a,c,e,f,d, high turbidity; ammonium chromate	Wallen et al. 1957[133]
Ammonia (NH_3)	212 ppm (2 day)	″	static acute bioassay; a,c,d,e,f, high turbidity, ammonium dichromate	″
	37 ppm (2 day)	″	static acute bioassay; a,c,d,e,g, highly turbid water; NH_4OH	″
	1,4000 ppm (2 day)	Gambusia affinis	static acute bioassay; a,c; ammonium sulfate; d,e,g; highly turbid water	Wallen et al. 1957[133]
	248 ppm (2 day)	Gambusia affinis	ammonium sulfide; static acute bioassay; a,c,d,e,g; highly turbid water used	Wallen et al. 1957[133]
	240 ppm (2 day)	″	same as above, but ammonium sulfite used.	″
	420 ppm (2 day)	″	static acute bioassay; a,c,d,e,g; Ammonium thiocyanate; highly turbid	″
	3.1 mg/l	Lepomis macrochirus	soft water; 30 C	Academy of Natural Sciences 1960[1]
	3.4 mg/l	″	soft water; 20 C	″
	23.7 mg/l	″	hard water; 30 C	″
	24.4 mg/l	″	hard water; 20 C	″
	90 mg/l	Physa heterostropha	soft water, 20 C	″
	94.5 mg/l	″	soft water, 30 C	″
	133.9 mg/l	″	hard water, 20 & 30 C	″
	6 mg/l	Lepomis macrochirus	In standard distilled water; NH_4Cl	″
	8.2 ppm	Pimephales promelas	static acute bioassay; in hard water; c,d,e,f	Henderson et al. 1960[60]
	5.2 ppm	Pimephales promelas	static acute bioassay; soft water; c,d,e,f.	″
	0.4 (24 hr)	Salmo gairdneri	unionized NH_3; static acute bioassay; a,b,c,d,e. toxicity increased with increasing pH (from 7.0 to 8.2)	Lloyd and Herbert 1960[76]
	24.6 ppm (2 day)	Salmo gairdneri	static acute bioassay, a,c,d,f NH_4Cl as N;	Herbert and Shurben 1964[53]
	202 ppm (1 day)	Carassius carassius	static acute bioassay; a,c, "standard reference water" NH_4Cl	Dowden and Bennett 1965[39]
	161 ppm (2 day)	″	″	″
	50 ppm	″	″	″
	139 ppm	Daphnia magna	″	″
	725 ppm (1–4 day)	Lepomis macrochirus	″	″
	241 ppm (1 day)	Lymnaea, sp (eggs)	″	″
	173 ppm (2 day)	″	″	″
	70 ppm	″	″	″
	60 ppm (1 day)	Daphnia magna	a,c; NH_4OH; static acute bioassay; "standard reference water"	″
	32 ppm (2 day)	″	″	″
	20 ppm	″	″	″
	423 ppm (1 day)	Daphnia magna	static acute bioassay, a,c, standard reference water and lake water using ammonium sulfate	″
	433 (2 day)	″	″	″
	292 ppm	″	″	″
	299 ppm (1 day)	″	static acute bioassay; a,c, standard reference water; ammonium sulfite	″
	273 ppm (2 day)	″	″	″
	203 ppm	″	″	″
	200 mg/l (4 days)	Cyprinus carpio	$(NH_4)_2SO_4$	Malacea 1966[78]
	300 mg/l (4 days)	gudgeon	″	″
	160 mg/l (4 days)	Rhodeus sericeus	″	″
	73.4 mg/l (2 days)	Daphnia	″	″

* Citations are listed at the end of the Appendix. They can be located alphabetically within tables or by their superior numbers which run consecutively across the tables for the entire Appendix.

TABLE 1—Continued

Constituent	Acute dose 96 hr LC50	Species	Conditions	Literature citation*
Ammonia (NH_3)	0.4 ppm (7 days)	Abramis brama	a,c,d,e,f; continuous-flow bioassay	Ball 1967[9]
	0.29 ppm (7 days)	Perca fluviatilis	"	"
	0.35 ppm (5 days)	Rutilis rutilis	"	"
	0.36 ppm (6 days)	Scardinius erythrophthalmus	"	"
	0.41 ppm (2 day)	Salmo gairdneri	"	"
	34–47 ppm (2 days)	Salmo gairdneri	acute static bioassay, a,c,d,e,f,o	Brown 1968[17]
	6.3 mg/l (48 hr)	Salmo gairdneri	ammonia as N	Brown 1968[17]
	420 ppm (5 day)	Nitzschia linearis	ammonium salt; a,c,e; static acute bioassay	Patrick et al. 1968[91]
	90.0 ppm	Physa heterostropha	"	"
	3.4 ppm	Lepomis macrochirus	ammonium salt; a,c,e; static acute bioassay	"
	0.44 ppm (3 hr)	Salmo gairdneri	100%; mortality un-ionized NH_3; 10.5 C pH 8–10	Lloyd and Orr 1969[77]
Antimony (Sb)	12 ppm	Pimephales promelas	antimony potassium tartrate; static acute bioassay; a,c,d,f; hard water	Tarzwell and Henderson 1960[124]
	20 ppm	"	same as above using soft water	"
	17 ppm	"	same as above, using hard water and antimony trichloride	"
	9 ppm	"	same as above except using soft water and antimony trichloride	"
	80 ppm	"	same as above using hard water and antimony trioxide	"
	80 ppm	"	same as above using soft water and antimony trioxide	"
(See sodium arsenite also)				
Arsenic (As)	48 ppm (24 hr)	Notropsis hudsonius		Boschetti and McLoughlin 1957[15]
	29 ppm (48 hr)	"		"
	27 ppm (72 hr)	"		"
		young salmon & trout	arsenic trioxide	Holland et al. 1960[57]
	100 mg/l (4 days)	Rhodeus sericeus	sodium arsenate	Malacea 1966[78]
	160 mg/l (4 days)	Cyprinus carpio	"	"
	5 mg/l (2 days)	Daphnia	"	"
Barium (Ba)	2083 mg/l (36 hr)	young eels	20 C; using $BaCl_2$	Doudoroff and Katz 1953[37]
	200 ppm (time not given)	Crassius auratus	$BaCl_2$	Bijan and Deschiens 1956[12]
	100 ppm (time not given)	Bulinus contortus	"	"
	11 ppm (time not given)	Planorbis glabratus	$BaCl_2$	Bijan and Deschiens 1956[12]
	1640 mg/l	Gambusia affinis	turbid water; 20 C $BaCl_2$	Wallen et al. 1957[133]
	4440 mg/l (24 hr)	"	"	"
	10,000 ppm (2 day)	Gambusia affinis	static acute bioassay, a,c,d,e,g; turbid water; barium carbonate; 20 C	Wallen et al. 1957[133]
	3,200 ppm (2 day)	"	same as above using barium chloride	"
Beryllium (Be)	1.3 mg/l	Lepomis macrochirus	beryllium sulfate; in soft water	Tarzwell and Henderson 1956[123]
	12 mg/l	"	", in hard water	"
	15 ppm	Pimephales promelas	static acute bioassay; a,c,d,f, hard water; beryllium chloride	"
	0.15 ppm	"	same as above using soft water	"
	20 ppm	"	same as above using hard water & beryllium nitrate	"
	0.15 ppm	"	same as above but using soft water	"
Beryllium (Be)	11 ppm	"	same as above but using hard water and beryllium sulfate	"
	0.2 ppm	"	same as above but using soft water	"
	31.0 mg/l	Fundulus heteroclitus	20–22 C; no feeding during the 96 hrs; aerated water	Jackim et al. 1970[64]
(See sodium borate, also)				
Boron (B)	15,000 mg/l (24 hr)	Lepomis macrochirus	20 C; borontifluoride	Turnbull et al. 1954[130]
	18,000–19,000 mg/l (6 hr)	minnows	in distilled water; 20 C;	LeClerc and Devaminck 1950[72], 1955[73]
	19,000–19,500 mg/l (6 hr)	"	in hard water; 20 C;	"
	18,000 mg/l (24 hr)	Gambusia affinis	boric acid; 20–23 C; pH 5.4–7.3	Wallen et al. 1957[133]
	5,600 mg/l	"	"	"
	12,000 mg/l (24 hr)	"	sodium borate, 22–26 C; pH 8.6–9.1	"
	8,200 mg/l (48 hr)	"	"	"
	3,600 mg/l	"	"	"
	10,500 ppm (2 day)	Gambusia affinis	boric acid; static acute bioassay; a,c,d,e,g; highly turbid water 20–23 C	Wallen et al. 1957[133]
Cadmium (Cd)	45 mg/l	Orizias	$Cd(NO_3)_2 \cdot 4H_2O$	Doudoroff and Katz 1953[37]
	0.056 mg/l	guppy	conc. as Cd; using $Cd(NO_3)_2 \cdot 4H_2O$	Shaw and Lowrance 1956[112]
	5 ppm	Pimephales promelas	static acute bioassay; a,c,d,f; hard water; cadmium chloride	Tarzwell and Henderson 1960[124]
	0.9 ppm	"	same as above, using soft water	"
	1.05 mg/l	"	static acute bioassay, c,d,e,f; soft water; $CdCl_2$; conc. as Cd.	Pickering and Henderson 1965[93]
	72.6 mg/l	"	same as above; using hard water	"
	1.94 mg/l	Lepomis macrochirus	static acute bioassay; c,d,e,f; soft water $CdCl_2$ conc. as Cd.	"
	1.27 mg/l	Lebistes reticulatus	same as above	"
	2.84 mg/l	Lepomis cyanellus	same as above	"
	66.0 mg/l	"	same as above, but using hard water	"
	0.17 ppm	Pimephales promelas	cadmium cyanide complex, sodium cyanide (439 ppm CN) and cadmium sulfate (528 ppm Cd). Synthetic soft water; static acute bioassay; a,c; conc. as CN	Doudoroff et al. 1966[38]
	0.008–0.01 ppm (7 day)	Salmo gairdneri	continuous flow, acute bioassay; a,b,f; hard water	Ball 1967[9]
	30 mg/l (1 day)	"	"	"
	30 ppm (1 day)	"	continuous flow, acute bioassay, a,b,f	Velsen and Alderdice 1967[132]
	0.12 mg/l (4–8 weeks)	Crassostrea virginica	in flowing water; 20 C salinity 31 ppt; $CdCl_2 \cdot 2.5H_2O$	Shuster and Pringle 1969[113]
	27.0 mg/l	Fundulus heteroclitus	20–22 C; no feeding during the 96 hr aerated water.	Jackim et al. 1970[64]
	0.2 mg/l (8 wk)	Crassostrea virginica		Pringle (in press)[99]
	0.1 mg/l (15 wk)	"		"
Calcium (Ca)	8,400 mg/l (24 hr)	Lepomis macrochirus		Doudoroff and Katz 1953[37]
	10,000 mg/l	Lepomis macrochirus	20 C; $Ca(NO_3)_2$	Trama 1954b[127]
	10,000 ppm	"	$Ca(NO_3)_2$; static acute bioassay; a,d,e,f	"

TABLE 1—Continued

Constituent	Acute dose 96 hr LC50	Species	Conditions	Literature citation*
Calcium (Ca)	10,650 ppm	"	$CaCl_2$; a,d,e,f; static acute bioassay in standard water	" Academy of Natural Sciences 1960[2]
	9,500 ppm	"	continuous flow, acute bioassay, a;c;ef; aerated water; small fish used.	Cairns Jr. and Scheier unpublished 1955,[142] 1958,[26] 1959[27] Industrial Wastes 1956[61]
	11,300 ppm	"	same as above except large fish used	Industrial Wastes 1956[61]
	7,752 mg/l (22–27 hr)	Carassius auratus	in distilled water	Jones 1957[67]
	160 mg/l	Gambusia affinis	$Ca(OH)_2$	Wallen et al. 1957[133]
	56,000 ppm	"	$CaCO_3$; a,c,d,e,g, turbid water static acute bioassay 19–21 C	"
	13,400 ppm (2 day)	"	$CaCl_2$; turbid water; static acute bioassay; a,c,d,e,g	"
	220 ppm (2 day)	"	$Ca(OH)_2$; a,c,d,e,g; static acute bioassay, turbid water; 21–23 C	"
	240 ppm (1 day)	"	"	"
	56,000 ppm (2 day)	"	$CaSO_4$; a,c,d,e,g; turbid water; static acute bioassay. 21–25 C	"
	11,300 ppm	Lepomis macrochirus	a,c,d,e,i; aerated water; $CaCl_2$ large fish used ~14.24 cm long; static acute bioassay.	Cairns Jr. and Scheier 1957,[28] 1958[26]
	saturation	"	18–20 C; in soft water; $CaSO_4$	Academy of Natural Sciences 1960[2]
	5% (time not given)	Gambusia affinis	20–23 C; DO 0.18–0.27 ppm (CO_2 = 13.75–69.30 ppm) $CaCl_2$	Ahuja 1964[3]
	3,525 ppm (1 day)	Daphnia magna	$CaCl_2$; a,c, static acute bioassay; standard reference water	Dowden and Bennett 1965[29]
	3,005 ppm (2 day)	"	same as above	"
	8,350 ppm (1 day)	Lepomis macrochirus	same as above	"
	4,485 ppm (1 day)	Lymnaea sp. (Eggs)	same as above	"
	3,094 (2 day)	"	same as above	"
	2,373 ppm (3 day)	"	same as above	"
	3,200 ppm (5 day)	Nitzschia linearis	static acute bioassay; a,c,e; $CaSO_4$	Patrick et al. 1968[91]
	2,980 ppm	Lepomis macrochirus	same as above	"
	3,130 ppm (5 day)	Nitzschia linearis	static acute bioassay; a,c,e, $CaCl_2$	Patrick et al. 1968[91]
	10,650 ppm	Lepomis macrochirus	same as above	"
(See also potassium chloride and sodium chloride)				
Chloride (Cl)	0.08 ppm (7 day)	Salmo gairdneri	continuous flow acute bioassay; a,c,e; from mono and dichloramines. 20 C; 23‰ salinity pH 8.0	Merkens 1958[81]
	10 ppm (24 hr)	Sphaerodes maculatus		Eisler and Edmunds 1966[11]
	19.25 ppm (16 hr)	fingerling silvers	conc. as residual Cl	Holland et al. 1960[57]
(See potassium chromate and dichromate and sodium chromate and dichromate also)				
Chromium (Cr)		Lepomis macrochirus	22±0.2 C	Abegg 1950[1]
	300 mg/l (24 hr)	"	Na_2CrO_4	Abegg 1950[1]
	145 mg/l (24 hr)	"	$Na_2Cr_2O_7$	"
	213 mg/l (48 hr)	"	$Na_2Cr_2O_7$	Turnbull et al. 1954[130]
	82 mg/l	Gambusia affinis	turbid water, 19–23 C; pH 7.5–7.8; 240 mg/l ammonium chromate	Wallen et al. 1957[133]
Chromium (Cr)	56 mg/l	"	turbid water, 18–20 C; pH 5.7–7.4; ammonium dichromate (136 mg/l)	"
	104 mg/l	"	turbid water; 17–21 C; pH 7.6–8.1 potassium chromate (400 mg/l)	"
	96 mg/l	"	turbid water; 21–30 C; pH 5.4–6.7 potassium dichromate (280 mg/l)	"
	135 mg/l	"	turbid water; 20–22 C; pH 7.7–8.6 sodium chromate; (420 mg/l)	"
	92 mg/l	"	turbid water; 24–27 C; pH 6.0–7.9 sodium dichromate (264 mg/l)	"
	103 mg/l	Lepomis macrochirus	$K_2Cr_2O_7$	Cairns Jr. and Scheier 1958;[26] 1959[27]
	40.0 ppm (48 hr)	"		"
	320 ppm	Lepomis macrochirus	in soft water; 18 C and 30 C	"
	382 ppm	"	in hard water; 18 C	"
	369 ppm	"	in hard water; 30 C	"
	196 mg/l (time not given)	Micropterus salmoides		Fromm and Schiffman 1958[42]
	110 ppm	Lepomis macrochirus	Cr hexavalent; static acute bioassay; a,c,d,f, g. soft water. alkali and hardness toxicity	Trama and Benoit 1958[128]
	110 mg/l	"	dichromate	Trama and Benoit 1958[128]
	170 mg/l	sunfish	$K_2Cr_2O_7$	Trama and Benoit 1958[128]
	100 mg/l	Salmo gairdneri	K_2CrO_4	Schiffman and Fromm 1959[110]
	113 mg/l	sunfish	$K_2Cr_2O_7$	Academy of Natural Sciences 1960[2]
	135 mg/l	sunfish	in hard water; $K_2Cr_2O_7$	"
	0.21 mg/l (time not given)	Navicula	22 C; time value; soft water	Academy of Natural Sciences 1960[2]
	0.25 mg/l (time not given)	"	22 C; "; hard water	"
	17.3 mg/l (time not given)	snail	time value; 20 C; soft water	Academy of Natural Sciences 1960[2]
	40.6 mg/l (time not given)	"	"; hard water	"
	110 mg/l	sunfish	$K_2Cr_2O_7$	Trama and Benoit 1960[129]
	75 mg/l (48 hr)	Polycelis nigra	20 C; hard water	Raymount and Shields 1962,[108] 1964[105]
	60 mg/l (12 days)	Carcinus maenas		"
	0.01 mg/l (48 hr)	Daphnia magna	chromic acid	Meletsea 1963[80]
	0.1 mg/l (48 hr)	"	potassium bichromate	"
	0.1 ppm (1 day)	Daphnia magna	chromic sulfate; a,c; standard reference water; static acute bioassay	Dowden and Bennett 1965[29]
	0.03 ppm (2 day)	"	same as above	"
	0.2 ppm (1 day)	Lymnaea sp (Eggs)	$Cr_2(SO_4)_3$ + $Na_2Cr_2O_7$; same as above	"
	5.07 mg/l	Pimephales promelas	chromium potassium sulfate; c,d,e,f soft water; static acute bioassay; conc. as Cr	Pickering and Henderson 1965[93]
	67.4 mg/l	"	same as above using hard water	"
	7.46 mg/l	Lepomis macrochirus	same as above using soft water	"
	71.9 mg/l	"	same as above using hard water	"
	4.10 mg/l	Carassius auratus	using soft water	"
		Lebistes reticulatus	same as above using soft water	"

TABLE 1—Continued

Constituent	Acute dose 96 hr LC50	Species	Conditions	Literature citation*
Chromium (Cr)	67.4–71.9 mg/l	Lepomis macrochirus	Hard water; pH 8.2 Alk. 300 mg/l	Pickering and Henderson 1966[94]
	3.33–7.46 mg/l	Pimelometopon pulchrum (fathead)	soft water; pH 7.5 Alk. 18 mg/l	"
	27.3–133 mg/l	minnows, Carassius auratus	hard water; pH 8.2 $K_2Cr_2O_7$; Alk. 300 mg/l	Pickering and Henderson 1966[94]
	17.6–118 mg/l	"	soft water; pH 7.5 $K_2Cr_2O_7$; Alk. 18 mg/l	"
	45.8 mg/l	"	soft water; pH 7.5 $K_2Cr_2O_7$; Alk. 18 mg/l	"
	180 ppm	zebra danio adults	conc. as Cr ($K_2Cr_2O_7$)	Cairns Jr. and Scheier 1968[27]
	1500 ppm	zebra danio eggs	"	"
	4.74 ppm	Lepomis macrochirus	a,c,d,e,; static acute bioassay fish acclimatized for 2 weeks in a synthetic dilution water using chromium-cyanide mixture	Cairns, Jr. and Scheier 1968[28]
	0.26 ppm	Lepomis macrochirus	a,c,d,e; static acute bioassay fish acclimatized for 2 weeks in a synthetic dilution water.	Cairns Jr. and Scheier 1968[28]
	170 ppm	Lepomis macrochirus	static acute bioassay, a,c,d,f,g; dichromate; fish were acclimated for 2 weeks in synthetic dilution.	Trama and Benoit 1958[125]
Copper (Cu)	1.0 ppm (6.5 day)	Gasterosteus aculeatus	static acute bioassay; a,c; using $Cu(NO_3)_2$	Jones 1938[65]
	0.23 mg/l (6 hr)	Balanus balanoides	hypertonic seawater	Pyefinch and Mott 1948[101]
	0.46 mg/l (6 hr)	" crenatus	"	"
	3.3 mg/l (24 hr)	Orizias	$CuCl_2 2H_2O$	Doudoroff and Katz 1953[37]
	0.74 ppm	Lepomis macrochirus	static acute bioassay; a,c,d,e. distilled aerated water	Trama 1954a[126]
	7.0 mg/l (48 hr)	"	20 C; pH [illegible]	Turnbull et al. 1954[130]
	0.18 ppm	Pimephales promelas	static acute bioassay; a,c,d,e,f; $CuSO_4$	[illegible] and Maloney 1955[90]
	84 ppm (2 day)	Gambusia affinis	static acute bioassay; turbid water; a,c,d,e,g; $CuSO_4$	Wallen et al. 1957[133]
	75 mg/l	Gambusia affinis	24–27 C; using copper sulfate in highly turbid water	Wallen et al. 1957[133]
	56,000 ppm (2 day)	Gambusia affinis	cupric oxide; static acute bioassay a,c,d,e,g; turbid water 19–20 C	Wallen et al. 1957[133]
	38 ppm (1 day)	Salmo gairdneri (fry)	$CuSO_4$; a,c,e,f,i,p; static acute bioassay	Turnbull-Kemp 1958[131]
	1.25 mg/l (time not given)	Lepomis macrochirus	in soft water; 18–20 C; $CuCl_2$	Academy of Natural Sciences 1960[2]
	48 hr.	Daphnia magna		Cabejszek and Stasiak 1960[22]
	1.9 ppm	Japanese oyster	Copper sulfate	Fujiya 1960[43]
	1.9 mg/l	oysters	pH 8.2; 12 C	Fujiya 1960,[43] 1961[44]
	1.4 ppm	Pimephales promelas	static acute bioassay; a,c,d,f, hard water; $CuSO_4$	Tarzwell and Henderson 1960[124]
	0.05 ppm	"	same as above using soft water	"
	10 ppm	Lepomis macrochirus	same as above using hard water	"
	0.2 ppm	"	same as above using soft water	"
	1.9 mg/l	oysters	$CuCl_2$ 2 H_2O	Fujya 1961[44]
	0.40 ppm	Limnodrilus hoffmeisteri	static acute bioassay; hard water $CuSO_4$; a,c,d,i	Wurtz and Bridges 1961[141]
	0.425 ppm	Gyraulus circumstriatus	static acute bioassay; a,c,d,i; hard water; $CuSO_4$	"
	0.27 ppm	Physa heterostropha	same as above	"
	1.5 mg/l (2–3 d)	Nereis	....	Raymount and Shields 1962[103]
	0.27 ppm	Physa heterostropha	21 C hard water as $CuSO_4$	Wurtz 1962[140]
	0.050 ppm	"	same as above; young	"
	0.56 ppm (1 day)	"	static acute bioassay; a,c,f, $CuSO_4$ hard and soft water.	Wurtz 1962[140]
	90 ppm (1 day)	Carassius auratus Poecilia reticulata	conc. as copper sulfite	Floch et al. 1963[41]
	15 ppm (1 day)	toad and frog tadpoles	Conc. as copper sulfite	Floch et al. 1963[41]
	10 ppm (2 days)	"	"	"
	5 ppm (3 days)	"	"	"
	20 ppm (3 day)	Dragon fly larvae	conc. as copper sulfite	Floch et al. 1963[41]
	40 ppm (1 day)	"	"	"
	2 ppm (1 day)	Daphnia longispina	"	"
	0.1 ppm	Nereis virens	time not specified	Raymount and Shields 1963[104]
	2 ppm (2 hr)	Salmo gairdneri	$CuSO_4 \cdot 5H_2O$	Herbert and Van Dyke 1964[54]
	0.2 ppm (48 hr)	"	"	"
	1.5 ppm	Gammarus lacustris	static acute bioassay; a,e, $CuSO_4$	Nebeker and Gaufin 1964[86]
	.19 ppm (12 days)	Nereis virens	time not specified	Raymount and Shields 1963[104]
	0.980 ppm	Lepomis macrochirus	$CuCl_2$	Cope 1965[31]
	2.8 ppm	Lepomis macrochirus	static acute bioassay; $CuSO_4$; a	"
	0.8 ppm (2 day)	Salmo gairdneri	a,c,e,f,l,m; field study in a river	Herbert et al. 1965[52]
	0.034 ppm (1 day)	Salmo salar	continuous flow, acute bioassay g,c,f; with 3 μg/l Zn and 2 μg/l Cu	Sprague 1965[118]
	32 μg/l (time not given)	juvenile salmon	in very soft water (14 mg/l hardness)	Sprague and Ramsay 1965[119]
	0.150 ppm (2 day)	Salmo gairdneri	static acute bioassay, a, $CuSO_4$	Cope 1966[32]
	2.800 ppm (2 day)	Lepomis macrochirus	same as above	Cope 1966[32]
	1.5 ppm	Pimephales promelas	as CN^- using copper cyanide complex; static acute bioassay; a,c; soft water	Doudoroff et al. 1966[38]
	1.2 ppm	"	same as above except conc. as Cu	"
	1.14 mg/l	Pimelometopon pulchrum	in hard water; $CuSO_4 \cdot 5H_2O$	Pickering and Henderson 1966[94]
	10.2 mg/l	Lepomis macrochirus	in hard water "	"
	0.048 ppm	Salmo salar	BSA;a; incipient lethal level with 0.600 Zn	Sigler et al. 1966[114]
	3.0 ppm	Orconectes rusticus	continuous flow acute bioassay, a,c,e,f; 20 C; intermolting stage	Hubschman 1967[58]
	1.0 ppm (1 day)	"	same as above; adult crayfish used	"
	1.0 ppm (6 day)	"	same as above; juvenile crayfish used	"
	1.0 ppm (6 day)	"	same as above; recently hatched young which remained clinging to pleopods of female during 1st molt were used.	"
	0.25 mg/l	Oroconectes rusticus embryo	time not given	Hubschman 1967[58]

TABLE 1—Continued

Constituent	Acute dose 96 hr LC50	Species	Conditions	Literature citation*
Copper (Cu)	0.57 mg/l (2 hr)	Wattersipora	copper sodium citrate pH 7.0–8.2	Wisely and Blick 1967[137]
	3.85 mg/l (2 hr)	Bugula	copper sodium citrate pH 7.0–8.2	Wisely and Blick 1967[137]
	0.51 mg/l (2 hr)	Spirorbis	copper sodium citrate pH 7.0–8.2	″
	2.9 mg/l (2 hr)	Galeolaria	coper sodium citrate pH 7.0–8.2	″
	22.5 mg/l (2 hr)	Mytilus	copper sodium citrate pH 7.2–8.2	″
	0.4–0.5 ppm (2 day)	Salmo gairdneri	static acute bioassay; a,c,d,e,f	Brown 1968[17]
	1.25 ppm	Lepomis macrochirus	static acute bioassay, a,c,d,e; Cu^{++}; fish acclimatized 2 wks. in syn. dil. water.	Cairns Jr. and Scheier 1968[25]
	1.04 ppm	″	static acute bioassay; a,c,d,e; fish acclimatized 2 wks in syn. dil. water copper-	″
	26.0 ppm	Lepomis macrochirus	copper acetic acid; all fisn acclimatized 2 wks. in syn. dil. water.	Cairns Jr. and Scheier 1968[25]
	5.2 ppm	″	a,c,d,e; static acute bioassay same as above except that copper-acetaldehyde was used.	″
	5.2 ppm	″	same as above except that acetone; copper mixture was used	″
	430 mg/l	adult minnows	static test	Mount 1968[93]
	470 mg/l	Pimephales promelas	continuous flow bioassay	Mount 1968[93]
	84.0 μg/l	Pimelometopon pulchrum	soft water; static bioassay	Mount and Stephen 1969[94]
	75 μg/l	″	″ continuous flow bioassay	″
	0.795–0.815 ppm (5 day)	Nitzschia linearis	static acute bioassay, a,c,e; $CuCl_2$	Patrick et al. 1968[91]
	1.25 ppm	Lepomis macrochirus	same as above	″
	0.2 mg/l (48 hr)	Penaeus duorarum	in the dark; 15 C; $CuSO_4$	Portmann 1968[97]
	30 mg/l (48 hr)	Penaeus aztecus	″	″
	100 mg/l″	shore crab	″	″
	1 mg/l″	cockle	″	″
	430 μg/l	Pimelometopon pulchrum	static bioassay; hard water	Mount and Stephen 1969[94]
	470 μg/l	″	continuous flow bioassay; hard water	Mount and Stephen 1969[94]
	1.7 mg/l	Capeloma decisum	soft water	Arthur and Leonard 1970[5]
	0.039 mg/l	Physa integra	soft water	Arthur and Leonard 1970[5]
	0.20 mg/l	Gammarus pseudolimnaeus	soft water	″
	48 hr	Salmo gairdneri		Brown and Dalton 1970[18]
	3.2 mg/l	Fundulus heteroclitus	20–22 C; no feeding during the 96 hrs. aerated water	Jackim et al. 1970
(See also potassium and sodium cyanides.)				
Cyanide (CN^-)	0.3 ppm (5.25–7.5 hr.)	Rhinichthys atratulus and Semotilus atromaculatus	ferro- and ferricyanides used. Conc. as cyanide used; daylight	Burdick and Lipscheutz 1948[21]
	0.33 mg/l (14 min.)	Coregonus artedii adult		Wuhrmann and Woker 1948[139]
	0.18 mg/l (24 hr)	Lepomis macrochirus	in soft water	Turnbull et al. 1954[130]
	0.06 ppm (1 day)	Lepomis auritus	continuous flow and static acute bioassays; a	Renn 1955[106]
	0.01–0.06 ppm (<1 day)	Lepomis macrochirus	same as above; static only	″
	0.05–0.06 ppm (<1 day)	″	same as above; continuous flow	″
Cyanide (CN^-)	0.06 ppm <1 day	Micropterus salmoides	static acute bioassay, a	″
	0.05–0.07 ppm (<1 day)	Pomotis annularis	static acute bioassay; a	″
	0.02–0.04 ppm (<1 day)	Pomotis annularis	continuous flow bioassay; a	″
	0.25 ppm (24 hr)	Pimephales promelas	NaCN; conc. as CN; 20 C	Doudoroff et al. 1966[38]
	0.24 ppm (48 hr)	″	″	″
	0.23 ppm	″	″	″
	0.20 ppm (24 hr)	″	conc. as CN^-; NaCN; plus 0.14 ppm Zn	″
	0.19 ppm (48 hr)	″	conc. as CN^-; NaCN; plus 0.13 ppm Zn	″
	0.18 ppm	Pimephales promelas	conc. as CN^-; NaCN	″
	0.23 ppm (24 hours)	″	″ plus 0.12 ppm Cd	″
	0.21 ppm (48 hr)	″	″ plus 0.11 ppm Cd	″
	0.17 ppm	″	″ plus 0.09 ppm Cd	″
	0.2 mg/l (11 min)	Salmo gairdneri		Neil 1957[87]
	0.12–0.18 mg/l	Lepomis macrochirus	in hard water and soft soft water	Academy of Natural Sciences 1960[2]
	0.16 mg/l	″	conc. as HCN	Doudoroff et al. 1966[38]
	0.01 mg/l (48 hr)	Salmo gairdneri		Brown 1968[17]
	0.18 ppm	Lepomis macrochirus	static acute bioassay; a,c,d,e; all fish acclimatized 2 weeks in syn. dil. water	Cairns Jr. and Scheier 1968[20]
	0.026 ppm	″	all fish acclimatized 2 weeks in syn. sil. water; static acute bioassay; a,c,d,e	″
	0.019 ppm	″	same as above; CN-Cr complex used	″
	4.74 ppm	″	same as above; CN-napthenic acid mixture	″
	0.026 ppm	″	same as above; CN used	″
	3.90 ppm	″	same as above; CN-Zn complex used	″
	0.432 ppm	Physa heterostropha	static acute bioassay; a,c,e	Patrick et al. 1968[91]
	0.18 ppm	Lepomis macrochirus	asme as above	″
(See also Manganese (Mn))				
Fluorine (F)	64 mg/l (10 days)	fish	using KF	Tarzwell 1957[122]
	2.7–4.6 mg/l (218 hrs)	Salmo gairdneri	using NaF; 55 C; 3.0 ppm Ca	Neuhold and Sigler 1960[88]
	75–91 mg/l	Cyprinus carpio	using NaF; 3 ppm Ca and Mg; 65–75 F	″
	222–273 ppm (424 hrs)	Salmo gairdneri	3 ppm Ca and Mg; 46 F	″
	242–261 ppm (214 hrs)	″	3 ppm Ca and Mg; 55 F	″
	2.3–7.3 mg/l (time not given)	Salmo gairdneri	18 C; in soft water using NaF	Angelovic et al. 1967[7]
	2.6–6.0 mg/l (time not given)	″	13 C; in soft water using NaF	″
	5.9–7.5 mg/l (time not given)	″	7.5 C; in soft water using NaF	″
Gold (Au)	0.40 mg/l (time not given)	stickleback		Jones 1939[66]
Iron (Fe)	74 ppm (2 day)	Gambusia affinis	static acute bioassay; a,c,d,e,g high turbidity; $FeCl_3$	Wallen et al. 1957[133]
	133 ppm (2 day)	Gambusia affinis	static acute bioassay; $Fe_2(SO_4)_3$ a,c,d,e,g; turbid water; 19–23 C	″
	10,000 ppm (2 day)	″	static acute bioassay; Fe_2O_3; a,c,d,e,g; turbid water; 16–23 C	″

TABLE 1—Continued

Constituent	Acute dose 96 hr LC50	Species	Conditions	Literature citation*
Iron (Fe)	10,000 ppm (2 day)	"	static acute bioassay; a,c,d,e,g; FeS; turbid water; 20–26 C	Wallen et al. 1957[133]
	350 ppm (2 day)	"	same as above except comp'd used was FeSO$_4$. 20–21 C.	"
	36 ppm (1 day)	Daphnia magna	static acute bioassay; a,c, standard ref. water; FeCl$_3$	Dowden and Bennet 1965[34]
	21 ppm (2 day)	"	"	"
	15 ppm	"	"	"
		mayflies, stoneflies, caddisflies	constant O$_2$; pH and hardness	Warnick and Bell 1969[134]
Lead (Pb)	0.3 ppm (4½ days)	Gasterosteus aculeatus	static acute bioassay; a,c, using Pb(NO$_3$)$_2$	Jones 1938[65]
	1.4 mg/l (48 hr)	Lepomis macrochirus	in tap water	Turnbull et al. 1954[130]
	2.0 mg/l (24 hr)	"	"	"
	6.3 mg/l (24 & 48 hrs)	"	"	"
	10 mg/l (24 & 48 hrs)	"	"; Pb(NO$_3$)$_2$	"
	240 mg/l	Gambusia affinis	Pb(NO$_3$)$_2$ used in highly turbid water	Wallen et al. 1957[133]
	75 mg/l	Pimephales promelas	in hard water	Tarzwell and Henderson 1956,[123] 1960[124]
	3.2 mg/l	"	in soft water; PbCl$_2$ used	"
	>100 mg/l	"	in hard water; PbCl$_2$ used	"
	26 mg/l (time not given)	Carassius auratus	PbSO$_4$ used	Jones 1957[67]
	240 ppm (2 days)	Gambusia affinis	static acute bioassay; a,c,d,e, turbid water Pb(NO$_3$)$_2$	Wallen et al. 1957[133]
	56,000 ppm (2 day)	Gambusia affinis	static acute bioassay; a,c,d,e,g; PbO; high turbidity	"
	0.34 mg/l (48 hr)	stickleback, Oncorhynchus kisutch	1000–3000 mg/l of dissolved solids	Gill et al. 1960[47]
	0.41 mg/l (24 hr)	Oncorhynchus kisutch	1000–3000 mg/l dissolved solids	"
	0.53 mg/l (24 hr)	sticklebacks	1000–3000 mg/l dissolved solids	"
	>75 ppm	Pimephales promelas	static acute bioassay; a,c,d,f, hard water; PbCl$_2$	Tarzwell and Henderson 1960[124]
	2.4 ppm (24 hrs)	"	same as above using soft water	"
	7.48 mg/l	Pimephales promelas & Lepomis macrochirus	static acute bioassay; c,d,e,f, soft water; lead acetate 7.8 mg/l DO; 18 mg/l Alk; 20 mg/l hardness	Pickering and Henderson 1965,[93] 1966[94]
	5.58 mg/l	Pimephales promelas	static acute bioassay; c,d,e,f; conc. as Pb; PbCl$_2$ used, soft water	Pickering and Henderson 1965[93]
	482.0 mg/l	"	same as above with hard water	"
	23.8 mg/l	Lepomis macrochirus	same as above with soft water	"
	442.0 mg/l	"	same as above with hard water	"
	31.5 mg/l	Carassius auratus	same as above with soft water	"
	20.6 mg/l	Lebistes reticulatus	same as above with soft water	"
	49.0 ppm (1 day)	tubificid worms	static acute bioassay; a,c; Pb(NO$_3$)$_2$ pH 6.5	Whitley 1968[135]
	27.5 ppm (1 day)	tubificid worms	"	"
	27.5 ppm (1 day)	tubificid worms	static acute bioassay; a,c, Pb(NO$_3$)$_2$	Whitley 1968[135]
	3.12 mg/l	Salvelinus fontinalis Fundulus heterocitus		Dorfman and Whitworth 1969[36]
	1–3 ppm (48 hrs)	Salmo gairdneri		Kariya et al. 1969[69]
Lead (Pb)		stoneflies, mayflies	O$_2$; pH and hardness are constant	Warnick and Bell 1969[134]
	188.0 mg/l	Fundulus heteroclitus	20–22 C; no feeding during 96 hours; aerated water	Jackim et al. 1970[64]
Magnesium (Mg)	16,500 mg/l	Gambusia affinis	in turbid water; MgCl$_2$·6H$_2$O	Wallen et al. 1957[133]
	17,750 ppm (2 day)	Gambusia affinis	BSA; a,c,d,e,g; turbid water MgCl$_2$	"
	15,500 ppm (2 day)	Gambusia affinis	BSA; a,c,e,d,g; turbid water MgSO$_4$	"
	3,391 ppm (1 day)	Daphnia magna	BSA; a,c, standard reference water; MgCl$_2$	Dowden and Bennett 1965[34]
	3,489 ppm	"	"	"
	2,101 ppm	"	BSA; a,c; standard reference water; MgSO$_4$	Dowden and Bennett 1965[34]
	19,000 ppm (1 day)	Lepomis macrochirus	same as above	"
	10,530 ppm (1 day)	Lymnaea sp. (eggs)	same as above	"
(See also Potassium permanganate)				
Manganese (Mn)	5500 mg/l (24 hrs)	fish, young eels	MnCl$_2$	Iwao 1936[62] Oshima 1931[89]
	500 mg/l (48 hrs)	Tinca tinca	MnF$_2$	Simonin and Pierron 1937[115]
	1000 mg/l (time not given)	fish	MnSO$_4$·H$_2$O; conc. as Mn	Meinck et al. 1956[79]
	7,850 mg/l (24 hrs)	Orizias	MnCl$_2$	Doudoroff and Katz 1953[37]
	1,400 ppm	Cyprinus carpio, killifish, Daphnia, Salmo gairdneri		Tabata 1969[121]
Mercury (Hg)	5 mg/l (75 hr)	Artemia salina	conc. as Hg using HgI$_2$; pH 8.1	Corner and Sparrow 1956[33]
	0.05 mg/l (2.5 hr)	Acttia clausi	conc. as Hg pH 8.1	Corner and Sparrow 1956[33]
	0.30 mg/l (")	Elminius	conc. as Hg pH 8.1	"
	800 mg/l (")	Artemia	conc. as Hg pH 8.1	"
	40 mg/l (22 hr)	Artemia salina	conc. as Hg using HgCl$_2$ pH 8.1	"
	0.9–60 mg/l	phytoplankton		Hueper 1960[59]
	0.027 mg/l (time not given)	bivalve larvae	HgCl$_2$ (0.02 mg/l of Hg)	Woelke 1961[138]
	0.04 mg/l	Rhodeus sericeus		Malacea 1966[78]
	0.05 mg/l	gudgeon		Malacea 1966[78]
	0.30 mg/l	Cyprinus carpio		"
	0.02 mg/l	minnow		"
	0.15 mg/l (48 hr)	Daphnia		"
	2.6 ppm (24 hr)	Ambassis safgha	conc. as HgCl$_2$	Ballard and Oliff 1969[10]
	6.5×10^{-6} M	Mytilus eduols planulatus	pH 7.8–8.2; HgCl$_2$	Wisely and Blick 1967[137]
	9.0×10^{-6} M	Crassostrea commercialis	pH 7.8–8.2; HgCl$_2$	"
	5.0×10^{-7} M (2 hours)	Wattersipora cucullata	pH 7.8–8.2; HgCl$_2$	Wisely and Blick 1967[137]
	1.0×10^{-7} M (2 hours)	Buluga neritina	pH 7.8–8.2; HgCl$_2$	Wisely and Blick 1967[137]
	7.0×10^{-7} M (2 hours)	Spirorbis lamellosa	pH 7.8–8.2; HgCl$_2$	"
	6.0×10^{-7} M (2 hours)	Galeolaria commercialis	pH 7.8–8.2; HgCl$_2$	"
	9.0×10^{-7} M (2 hours)	Artermia sanna	pH 7.8–8.2; HgCl$_2$	"
	0.1 mg/l (48 hr)	Penaeus duorarum	15; in the dark; HgCl$_2$	Portmann 1968[97]
	6 mg/l (48 hr)	Penaeus aztecus	15 C; in the dark; HgCl$_2$	"
	1 mg/l (48 hr)	Hemigrapsis oregonensis	"	"
	10 mg/l (48 hr)	Clinocardium nuttalli	"	"
	26 ppm (24 hr)	Daphnia magna Ambassia safgha	Lake Erie water in sealed containers; HgCl$_2$	Ballard and Oliff 1969[10]
	0.23 mg/l	Fundulus heteroclitus	20–22 C; no feeding during the 96 hrs; aerated water	Jackim et al. 1970[64]

TABLE 1—Continued

Constituent	Acute dose 96 hr LC50	Species	Conditions	Literature citation*
Molybdenum (Mo)	70 mg/l	Pimephales promelas	MoO$_3$; pH 7.4; Alk. 18 mg/l; hardness 20 mg/l; soft water	Tarzwell and Henderson 1956[123]
	370 mg/l	"	MoO$_3$; pH 8.2; Alk. 360 mg/l; hardness 400 mg/l; hard water	"
Nickel (Ni)	0.8 mg/l (time not given)	sticklebacks	concentration as Ni; Ni(NO$_3$)$_2$	Murdoch 1953[95]
	0.95 mg/l	Pimephales promelas	BSA; a,c,d; nickel cyanide complex syn. soft water	Doudoroff et al. 1966[39]
	1.0 mg/l (time not given)	sticklebacks	concentration as Ni; 15–18 C	Jones 1957[67]
	24 ppm	Pimephales promelas	BSA; a,c,d,f; hard water nickelous chloride	Tarzwell and Henderson 1960[124]
	4 ppm	"	same as above using soft water	"
	25 ppm (2 day)	Salmo gairdneri	field study on a river; a,c,e,f,l,m	Herbert et al. 1965[61]
	5.18 mg/l	Pimephales promelas	BSA; c,d,e,f; soft water; nickel chloride; conc. as Ni	Pickering and Henderson 1965,[97] 1966[91]
	42.4 mg/l	"	same as above using hard water	"
	5.18 mg/l	Lepomis macrochirus		"
	39.6 mg/l	"	same as above using hard water	"
	9.82 mg/l	Carassius auratus	same as above using soft water	"
	4.45 mg/l	Lebistes reticulatus	same as above using soft water	"
	160 ppm (2 day)	Salmo gairdneri	BSA; a,f; NiSO$_4$	Willford 1966[136]
	270 ppm (2 day)	Salmo trutta	BSA; a,f; NiSO$_4$	Willford 1966[136]
	242 ppm (2 day)	Salvelinus fontinalis	same as above	"
	75 ppm (2 day)	Salvelinus namaycush	same as above	"
	165 ppm (2 day)	Ictalurus punctatus	same as above	"
	495 ppm (2 day)	Lepomis macrochirus	same as above	"
	200 mg/l (48 hr)	Penaeus duorarum	15 C; in the dark; NiSO$_4$	Portmann 1968[99]
	150 mg/l (48 hrs)	Penaeus aztecus	"	"
	300 mg/l (48 hrs)	Hemigrapsus oregonensis	"	"
	500 mg/l (48 hrs)	Clinocardium nuttalli	"	"
	48 hrs	Salmo gairdneri		Brown and Dalton 1970[18]
See also sodium nitrate				
Nitrate (NO_3^-)	64 hours	Daphnia magna	25 C; Lake Erie water; daphnids 8-hours old	Anderson 1948[4]
	8.1 mg/l (24 hours)	Gambusia affinis	21–24 C; in highly turbid water; NaNO$_3$	Wallen et al. 1957[133]
	9.5 mg/l (48 & 96 hr)	"	"	"
pH	1.3 ppm (45 min)	Ictalurus punctatus	static acute bioassay; a,c, H$_2$S; Using advanced fingerlings	Bonn and Follis 1967[14]
	1.4 ppm (45 min)	"	same as above, using adults	"
	0.07 ppm (2 day)	Salmo gairdneri	static acute bioassay; HCN; a,c,d,e,f,o	Brown 1968[17]
	10 mg/l	trout		Belding 1927[11]
	pH 4.0 (time not given)	Carassius auratus		Jones 1939[66]
	pH 3.65 (3 day)	Lepomis macrochirus	continuous flow, acute bioassay HCl; a,c,e,f	Cairns Jr. and Scheier unpublished 1955[142]
	0.069 ppm (1 day)	Lagondon rhomboides	static acute bioassay; HCN; aerated sea water; a;	Daugherty and Garrett 1951[35]
	"	"	aerated sea water; static acute bioassay HCN	Garrett 1957[50]
pH	282 ppm (2 day)	Gambusia affinis	static acute bioassay; HCl; turbid water; a,c,d,e,g	Wallen et al. 1957[133]
	pH 10.5	Lepomis macrochirus	maximum pH	Cairns Jr. and Scheier 1958[26]
	pH 3.5	Lepomis macrochirus	a,c,d,e,i; dist. aerated water; large fish used; 14.24 cm length 20 C	Cairns Jr. and Scheier 1959[27]
	4–6 mg/l (6 hr)	minnows	in distilled water; HCl	LeClerc 1960[71]
	100–110 mg/l (6 hrs)	minnows	in hard water, HCl	"
	0.16 ppm (3 day)	Lepomis macrochirus	juveniles used; HCN; static acute bioassay; a,c,d,f,p;	Doudoroff et al. 1966
	1.0 ppm (20 min)	Ictalurus punctatus fingerlings	static acute bioassay; a,c, H$_2$S	Bonn and Follis 1967[14]
Phosphate (PO_4^{3-})	24 hours	Lepomis macrochirus	22 ± 0.2 C	Abegg 1950[1]
	720 mg/l	Gambussia affinis	turbid water; 19–23 C; NaH$_2$PO$_4$	Wallen et al. 1957[133]
	1380 mg/l	"	turbid water; 19–24 C; Na$_4$P$_2$O$_7$·10H$_2$O	"
	151 mg/l	"	turbid water; 17–22 C; Na$_3$PO$_4$	"
	138 ppm (2 day)	"	BSA; a,c,d,e,g; turbid water phosphoric acid; 22–24 C	Wallen et al. 1957[133]
Phosphorus (P)	0.105 ppm (2 day)	Lepomis macrochirus	BSA; a,c,d,e,f,g,h,i,j,k, n,o; colloidal premoved; 26 C; conc. as P	Isom 1960[62]
	0.053 ppm (3 day)	"	same as above	"
	0.025 ppm	"	same as above	"
Potassium (K)	2.0 ppm (2 day)	Hydropsyche	BSA; a, soft water; KCN	Roback 1965[107]
	0.5 ppm (2 day)	Stenonema	"	"
	2010 ppm	Lepomis macrochirus	BSA; a,c,d,e,f; KCl	Trama 1954b[127]
	3,000 ppm	"	BSA; a,d,e,f; KNO$_3$; syn. dilution water	"
	450 ppm	"	a,c,e,f; aerated dist. water; K$_2$CrO$_4$; small fish; continuous flow acute bioassay	Cairns Jr. and Scheier unpublished 1955[142]
	630 ppm	"	continuous flow, acute bioassay K$_2$CrO$_4$; medium size fish a,c,e,f; pH 7.9 to 8.6	"
	5.50 ppm	"	same as above using large fish	"
	0.22 ppm (1 day)	Rhinichthys atratulus meleagris	continuous flow, acute bioassay; a,c,e; KCN	Lipschuetz and Cooper 1955[74]
	0.45 ppm	Lepomis macrochirus	BSA; a,c,e; KCN	Cairns Jr. 1957[2]
	320 ppm	"	BSA; a,c,e, K$_2$Cr$_2$O$_7$	Cairns Jr. 1957[2]
	4,200 ppm (2 day)	Gambusia affinis	BSA; a,c,d,e,g; turbid water KCl	Wallen et al. 1957[133]
	400 ppm (2 day)	"	BSA; a,c,d,e,g; highly turbid water; K$_2$CrO$_4$	Wallen et al. 1957[133]
	1.6 ppm (2 day)	"	BSA; a,c,d,e,g; KCN, turbid water	Wallen et al. 1957[133]
	320 ppm (2 day)	Gambusia affinis	BSA; a,c,d,e,g; turbid water; K$_2$Cr$_2$O$_7$	Wallen et al. 1957[133]
	324 ppm (2 day)	Gambusia affinis	BSA; a,c,d,e,g; turbid water KNO$_3$	Wallen et al. 1957[133]
	12 ppm (2 day)	Gambusia affinis	BSA; a,c,d,e,g; turbid water KMnO$_4$	Wallen et al. 1957[133]
	0.45 ppm	Lepomis macrochirus	BSA, a,e; KCN; 5–9 ppm oxygen	Cairns Jr. and Scheier 1958[26]
	0.12 ppm	"	BSA; a,e; KCN; 2 ppm DO	"
	1.08 ppm	Physa heterostropha	BSA; a,e; KCN; 5–9 ppm DO	"
	0.48 ppm	Physa heterostropha	BSA; a,e; KCN 2 ppm DO	"

TABLE 1—Continued

Constituent	Acute dose 96 hr LC50	Species	Conditions	Literature citation*
Potassium (K)	320 ppm	Lepomis macrochirus	continuous flow, acute bioassay, $K_2Cr_2O_7$; aerated distilled water pH 6.2; a,c,e,f;	Cairns Jr. and Scheier 1958[24]
	320 ppm	"	BSA; a,e,; 5-9 ppm DO	"
	320 ppm	"	" 2 ppm DO	"
	195 ppm	Micropterus salmoides	BSA; a,c,d,e; K_2CrO_4	Fromm and Schiffman 1958[47]
	1,337 ppm (5 day)	Nitzschia linearis	BSA; a,c,e; KCl	Patrick et al. 1968[91]
	940 ppm	Lepomis macrochirus	same as above	"
	2,010 ppm	Physa heterostropha	same as above	"
	7.8 ppm (5 day)	Nitzschia linearis	BSA; a,c,e; K_2CrO_4	Patrick et al. 1968[91]
	16.8 ppm	Physa heterostropha	same as above	Patrick et al. 1968[91]
	168.8 ppm	Lepomis macrochirus	same as above	"
	550 ppm	Lepomis macrochirus	BSA; a,c,d,e,i; aerated dist. water; K_2CrO_4; large fish used; 14-24 cm long	Cairns Jr. and Scheier 1959[27]
	0.57 ppm	Lepomis macrochirus	BSA; a,b,c,d,e,i; aerated distilled water; large fish used 14.24 cm in length; KCN	"
	320 ppm	"	BSA; a,c,d,e,f; 18-30 C; $K_2Cr_2O_7$	"
	382 ppm	"	same except in hard water at 18 C	"
	369 ppm	"	" at 30 C	"
	320 ppm	"	distilled aerated water; BSA; a,c,d,e,i; $K_2Cr_2O_7$; fish 14.24 cm	"
	100 ppm (1 day)	Salmo gairdneri	BSA; a,c,d,g; K_2CrO_4	Schiffman and Fromm 1959[110]
	0.43 ppm	Lepomis macrochirus	BSA; a,c,d,e,f; KCN	Cairns Jr. and Scheier unpublished 1955[142]
	0.45 ppm	"	BSA; a,e, KCN; normal oxygen content	Cairns Jr. 1965[21]
	0.12 ppm	"	BSA; a,e; KCN; low oxygen content	"
	1.08 ppm	Physa heterostropha	BSA; a,e; KCN; normal oxygen content in water	"
	0.4[illegible] ppm	"	BSA; a,e; KCN; low oxygen content	"
	320 ppm	Lepomis macrochirus	BSA; a,e; $K_2Cr_2O_7$; normal DO content in water	"
	320 ppm	"	BSA; a,e; $K_2Cr_2O_7$, low DO content in water	"
	0.49 ppm (2 days)	Brachydanio rerio	BSA; a,c,d,e,f; dist. water adults KCN; 24 C; 5-9 ppm	Cairns Jr. et al. 1965[22]
	117 ppm (2 day)	Brachydanio rerio	BSA; a,c,d,e,f; KCN; eggs 24 C; 5-9 ppm DO; distilled aerated water.	"
	0.16 ppm (2 day)	Lepomis macrochirus	same as above (not eggs)	"
	180 ppm (2 day)	Brachydanio rerio	BSA; a,c,d,e,f; $K_2Cr_2O_7$; 24 C; 5-9 ppm DO adults	"
	1500 ppm (2 day)	"	same as above using eggs	"
	440 ppm (2 day)	"	same as above not using eggs	"
	679 ppm (1 day)	Daphnia magna	BSA; a,c; standard ref. water KCl	Dowden and Bennett 1965[39]
	5,500 ppm (1 day)	Lepomis macrochirus	same as above	Dowden and Bennett 1965[39]
	1,941 ppm (1 day)	Lymnaea sp.	same as above	"
Potassium (K)	2 ppm (1 day)	Daphnia magna	BSA; a,c; standard reference water; KCN	Dowden and Bennett 1965[39]
	0.7 ppm (3 day)	"	BSA; a,c; standard reference water; KCN	"
	0.4 ppm	"	BSA; a,c; standard reference water; KCN	"
	796 ppm (1 day)	Lymnaea sp.	BSA; a,c; standard reference water; KCN	Dowden and Bennett 1965[39]
	147 ppm (3 day)	Lymnaea sp.	BSA; a,c; standard reference water; KCN	"
	130 ppm	"	"	"
	705 ppm (1 day)	Carassius carassius	BSA; a,c; standard reference water $K_2Cr_2O_7$	Dowden and Bennett 1965[39]
	0.4 ppm	Daphnia magna	same as above	"
	739 ppm (1 day)	Lepomis macrochirus	"	"
	905 ppm (1 day)	Daphnia magna	$K_3Fe(CN)_6$; BSA; a,c; standard reference water	Dowden and Bennett 1965[39]
	549 ppm (2 day)	Daphnia magna	same as above	"
	0.6 ppm (3 day)	"	same as above	"
	0.1 ppm	"	same as above	"
	900 ppm	"	BSA; a,c; KNO_3; standard reference water	"
	45.6 mg/l	Pimephales promelas	BSA; c,d,e,f; soft water; K_2CrO_4 conc. as Cr	Pickering and Henderson 1965[93]
	17.6 mg/l	Pimephales promelas	BSA; c,d,e,f; soft water; $K_2Cr_2O_7$ conc. as Cr	"
	27.3 mg	"	same as above using hard water	"
	118.0 mg/l	Lepomis macrochirus	same as above using soft water	Pickering and Henderson 1965[93]
	133.0 mg/l	Lepomis macrochirus	same as above using hard water	"
	37.5 mg/l	Carassius auratus	same as above using soft water	"
	30.0 mg/l	Lebistes reticulatus	same as above using soft water	"
	28.0 ppm (2 day)	Hydropsyche and Stenonema	BSA; a; soft water; $K_2Cr_2O_7$	Roback 1965[107]
	3.5 ppm (2 day)	"	"	"
	4.2 ppm (4 day)	Lepomis macrochirus	BSA; $KMnO_4$	Kemp et al. 1966[69]
	3.7 ppm	Semotilus atromaculatus	BSA; $KMnO_4$	"
	0.208 ppm	Nitzschia linearis	BSA; a;c;e; $K_2Cr_2O_7$	Patrick et al. 1968[91]
	17.3 ppm	Physa heterostropha	BSA; a,c; $K_2Cr_2O_7$	"
	113.0 ppm	Lepomis macrochirus	same as above	"
Selenium (Se)	2.5 mg/l	Daphnia	23 C; conc. as Se; added sodium selenite	Bringmann and Kuhn 1959[16]
Silver (Ag)	0.0043 mg/l (time not given)	guppies	conc. of Ag, placed in water as silver nitrate	Shaw and Lowrance 1956[112]
	0.04 mg/l	Fundulus heteroclitus	20-22 C; no feeding during the 96 hours; aerated water	Jackim et 1970[14]
Sodium (Na)	12,946 ppm	Lepomis macrochirus	static acute bioassay; a,d,e,f; synthetic dilution water; NaCl	Trama 1954b[127]
	12,000 ppm	Lepomis macrochirus	static acute bioassay; a,c,e,f, synthetic dilution water; $NaNO_3$ 20 C	Trama 1954b[127]
	0.23 ppm	Pimephales promelas	BSA; a,c, NaCN; syn. soft water; conc. as CN	Doudoroff et al. 1966[38]
	45 ppm (1 day)	Notropsis hudsonius	BSA, a,c,d,e; $NaAsO_2$	Boschetti and McLoughlin 1957[15]
	29 ppm (2 day)	Notropsis hudsonius	"	"
	27 ppm (3 day)	"	"	"
	8,200 ppm (2 day)	Gambusia affinis	static acute bioassay; a,c,d,e,g; $Na_2B_4O_7$ used; turbid water	Wallen et al. 1957[133]

TABLE 1—Continued

Constituent	Acute dose 96 hr LC50	Species	Conditions	Literature citation*
Sodium (Na)	18,100 ppm (2 day)	"	BSA; a,c,d,e,g; turbid water; using NaCl	"
	500 ppm (2 day)	Gambusia affinis	BSA; a,c,d,e,g; Na_2CrO_4; turbid water	"
	420 ppm (2 day)	"	BSA; $Na_2Cr_2O_7$; a,c,d,e,g; turbid water	"
	925 ppm (2 day)	"	BSA; a,c,d,e,g; NaF; turbid water	"
	10,000 ppm (2 day)	"	static acute bioassay; a,c,d,e,g; turbid water; $NaNO_3$	"
	2,400 ppm (2 day)	"	BSA; a,c,d,e,g; turbid water; $NaSiO_3$	"
	750 ppm (2 day)	"	BSA; a,c,d,e,g; Na_2S; turbid water	"
	9,500 ppm	Lepomis macrochirus	a,c,e,f; $NaNO_3$; aerated distilled water	Cairns Jr. and Scheier 1958,[26] 1959[27]
	9,000 ppm	Lepomis macrochirus	BSA; a,c,d,e,i; aerated water; distilled; $NaNO_3$; large fish	"
	0.35 ppm	Pimephales promelas	BSA; c,d,e,f; NaCN hard water	Henderson et al. 1959[59]
	0.23 ppm	"	same as above using soft water	"
	0.15 ppm	Lepomis macrochirus	same as above using hard water	Henderson et al. 1959[59]
	0.78 percent NaCl 72 hrs	Daphnia magna	NaCl at 25 C	Prasad 1959[99]
	0.93 percent NaCl (24 hrs)	Daphnia magna	NaCl at 25 C	Prasad 1959[99]
	0.50 percent NaCl (72 hrs)	Daphnia magna	NaCl at 50 C	Prasad 1959[99]
	5.9–7.5 ppm (2 days)	Salmo gairdneri	BSA; a; 45 F; NaF	Academy of Natural Sciences 1960[2]
	2.6–6.0 ppm (2 day)	"	BSA; a; 55 C; NaF	"
	6,200 ppm	Limnodrilus hoffmeister	BSA; a,c,d,i; NaCl	Wurtz and Bridges 1961[141]
	7,500 ppm	"		"
	6,150 ppm	Erpobdella punctata		"
	3,200 ppm	Helisoma campanulata	BSA; a,c,d,i; NaCl	"
	3,500 ppm	Gyraulus circumstristus		"
	5,100 ppm	Physa heterostropha		"
	6,200 ppm	Physa heterostropha	BSA; a,c,d,i; NaCl	Wurtz and Bridges 1961[141]
	1,100 ppm	Sphaerium cf. tenue		"
	1,150 ppm	"		"
	8,250 ppm	Asellus communis		"
	24,000 ppm	Argia sp.	same as above using hard water	"
	1.0 percent (36 mins)	Nais sp.	BSA; a,f; hard water; NaCl	Learner and Edwards 1963[70]
	60.0 ppm (2 day)	Carcinus maenas	BSA; a; Na_2CrO_4	Raymount and Shields 1962[103]
	26 ppm	Salmo gairdneri	BSA; a; $NaAsO_2$; 55–75 F	Cope 1965[31]
	30 ppm	Lepomis macrochirus	BSA; a; $NaAsO_2$; 55–75 F	"
	45 ppm	Pteronarcys	BSA; a; $NaAsO_2$; 60 F	Cope 1965[31]
	14,120 ppm (1 day)	Daphnia magna	static acute bioassay; a,c, standard reference water; conc. as $NaSiO_3$ plus 950 ppm $NaHSO_3$	Dowden and Bennett 1965[39]
	11,723 ppm (2 day)	Daphnia magna	same as above, but with 785 ppm $NaHSO_3$	"
	22 ppm	Daphnia magna	same as above, but with 15 ppm $NaHSO_3$	"
	0.15 ppm	"	BSA; a,c; standard reference water; conc. as Na_2CrO_4; plus 187 ppm Na_2CO_3 plus 88 ppm $NaSiO_3$	"
Sodium (Na)	0.19 ppm	Daphnia magna	BSA; a,c; standard ref. water; conc. as Na_2CrO_4; plus 240 ppm Na_2CO_3 plus 2,078 ppm Na_2SO_4	Dowden and Bennett 1965[39]
	76 ppm	Daphnia magna	BSA; a,c; standard ref. water; conc. as Na_2SiO_3; plus 161 ppm Na_2CO_3; plus 1,396 ppm Na_2SO_4	"
	9,000 ppm (2 day)	Hydropsyche	BSA; a; NaCl; soft water	Roback 1965[107]
	2,500 ppm (2 day)	Stenonema	"	"
	13,750 ppm (1 day)	Carassius carassius	BSA; a; c; NaCl; standard ref. water	Dowden and Bennett 1965[39]
	10,500 ppm (1 day)	Culex sp (larvae)	"	"
	6,447 ppm (1 day)	Daphnia magna	BSA; a,c; NaCl; standard ref. water	Dowden and Bennett 1965[39]
	14,125 ppm (1 day)	Lepomis macrochirus	"	"
	3,412 ppm (1 day)	Lymnaea sp (eggs)	"	"
	18,735 ppm (1 day)	Mollienesa latopinna	"	"
	0.21 ppm	Daphnia magna	BSA; a,c; standard ref. water; Na_2CrO_4; plus 130 ppm $NaSiO_3$	"
	0.28 ppm	"	same as above; conc. as Na_2CrO_4 plus 3,044 ppm Na_2SO_4	"
	22 ppm (1 day)	Daphnia magna	BSA; a,c; $Na_2Cr_2O_7$; standard ref. water	Dowden and Bennett 1965[39]
	4,206 ppm	Daphnia magna	BSA; a,c; standard reference water; $NaNO_3$	"
	12,800 ppm (1 day)	Lepomis macrochirus	BSA; a,c; $NaNO_3$; standard ref. water	"
	6,375 ppm (1 day)	Lymnaea sp. (eggs)	"	"
	5,950 ppm (2 day)	"	BSA; a,c; $NaNO_3$; standard ref. water	"
	[illegible] ppm	"	"	"
	895 ppm (1 day)	Amphipoda	BSA; a,c; $NaSiO_3$; standard ref. water	"
	630 ppm (1–4 days)	Lymnaea sp. (eggs)	BSA; a,c; $NaSiO_3$; standard ref. water	"
	16 ppm (1 day)	Daphnia magna	standard ref. water; Na_2S; a,c; BSA;	"
	13 ppm (2 day)	"	"	"
	9 ppm	"	"	"
	36.5 ppm (2 day)	Salmo gairdneri	static acute bioassay; a, $NaAsO_2$	Cope 1966[32]
	44.0 ppm (2 day)	Lepomis macrochirus	same as above	"
	80.0 ppm (2 day)	Pteronarcys californica	same as above	"
	1.8 ppm (2 day)	Daphnia magna	same as above	Cope 1966[32]
	1.4 ppm (2 day)	Simocephalus serrulatus	same as above	"
	44 ppm (LC50)	Lepomis macrochirus	BSA; a,c,d,i,g; $NaAsO_2$	Crosby and Tucker 1966[34]
	60 ppm (LC50)	Salmo gairdneri	BSA; $NaAsO_2$; a,c,d,i,g	Crosby and Tucker 1966[34]
	25 ppm	"	Field study-river; a,c,f,i,m; $NaAsO_2$	Gilderhus 1966[47]
	34 ppm	Carassius auratus	same as above	"
	35 ppm	Lepomis macrochirus	same as above	"
	0.038 ppm	Pteronarcys californica	static acute bioassay; a,c,d,e,f; $NaAsO_2$	Sanders and Cope 1966[109]
	2800 ppm (1 day)	Salmo gairdneri	static acute bioassay; a,e $Na_2B_4O_7$	Alabaster 1967[4]
	1800 ppm (2 day)	"	same as above	"
	0.7 ppm (1 day)	Lepomis macrochirus	static acute bioassay; a,b,e; $NaAsO_2$	Hughes and Davis 1967[60]
	2,430 ppm (5 day)	Nitzschia linearis	BSA; a,c,e; NaCl	Patrick et al. 1968[91]
	12,940 ppm	Lepomis macrochirus	BSA; a,c,e; NaCl	"

TABLE 1—Continued

Constituent	Acute dose 96 hr LC50	Species	Conditions	Literature citation*
Sulphide (See sodium sulphide and hydrogen sulphide under sodium and hydrogen (H⁺))				
Titanium (Ti)	120 ppm	Pimephales promelas	BSA; a,c,d,f; titanium sulfate; hard water	Tarzwell and Henderson 1956[123], 1960[124]
	8.2 ppm	"	same as above using soft water	"
Uranium (U)	3.7 ppm	Pimephales promelas	BSA; a,c,d,f; uranyl acetate soft water	Tarzwell and Henderson 1956[123], 1960[124]
	3.1 ppm	"	same as above using uranyl nitrate	"
	135 ppm	"	BSA; a,c,d,f; uranyl sulfate hard water	"
	2.8 ppm	"	same as above using soft water	"
Vanadium (V)	55 ppm	Pimephales promelas	BSA; a,c; vanadium pentoxide hard water	"
	13 ppm	"	same as above using soft water	"
	30 ppm	"	BSA; a,c; vanadyl sulfate hard water	"
	4.8 ppm	"	same as above using soft water	"
	55 ppm	Lepomis macrochirus	same as above using hard water	"
	6 ppm	"	same as above using soft water	"
Zinc (Zn)	0.7 ppm (4.5 days)	Gasterosteus aculeatus	BSA; a,c; zinc sulphate	Jones 1939[66]
	0.072 ppm (64 hr)	Daphnia magna	Lake Erie water; 25 C	Anderson 1948[6]
	2–6 ppm (24 hr)	Salmo gairdneri fingerlings	hard water	Goodman 1951[48]
	3–4 ppm (48 hr)	"	hard water	Goodman 1951[48]
	13 ppm	Biomphalaria biossyi	14 C; pH 7.8 ± 0.2; oxygenated tap water	Hoffman and Zakhary 1951[56]
	4.8 ppm	"	17 C; pH 7.8 ± 0.2; oxygenated tap water	"
	1.4 ppm	"	20 C; pH 7.8 ± 0.2; oxygenated tap water	"
	0.58 ppm	Biomphalaria biossyi	23 C; pH 7.8 ± 0.2; oxygenated	Hoffman and Zakhary 1951[56]
	0.6 ppm	Salmo gairdneri fingerlings	zinc chlorate and sulfate used. 17.5 C, diluted, well water used; 4.5 ppm Ca	Lloyd 1960[75]
	2.86–3.63 ppm	Lepomis macrochirus	18–30 C; soft water	Cairns Jr. and Scheier 1957[25]
	10–12 ppm	"	18–30 C; hard water	"
	7.20 mg/l	Lepomis macrochirus	standard dilution water; 20 C; $ZnCl_2$ concentration	Cairns Jr. and Scheier 1958,[26] 1959[27]
	3.5 mg/l	"	standard dilution water; 20 C	" 1958[26]
	8.02 mg/l	"	standard dilution water	"
	10–12 ppm	Lepomis macrochirus	static acute bioassay; a,c,d,e,f,i,n,g; 18 C; hard water	Cairns Jr. and Scheier 1958[26]
	2.9–3.8 ppm	adult	same as above using soft water	"
	10–12 ppm	Lepomis macrochirus	BSA; a,c,d,e,f,i; 30 C; hard water	Cairns Jr. and Scheier 1958[26]
	1.9–3.6 ppm	" (adult)	"; soft water	"
	8.02 ppm	Lepomis macrochirus	BSA; a,e; $ZnCl_2$; conc. as Zn^{+2} 5–9 ppm DO;	Cairns Jr. and Scheier 1958[26]
	4.9 ppm	"	same as above using 2 ppm DO	"
	0.79–1.27 ppm	Physa heterostropha	BSA; a,c,d,e,g; 20 C; Zn ion soft water	"
	2.66–5.57 ppm	"	same as above using hard water	"
	0.62–0.78 ppm	"	BSA; a,c,d,e,g; 30 C; Zn ion soft water	"
	2.36–6.36 ppm	"	same as above using hard water	"
Zinc (Zn)	6.91 ppm	Lepomis macrochirus	BSA; a,c,d,e,i; $ZnCl_2$; conc. as Zn + 2; aerated distilled water; large fish	Cairns Jr. and Scheier 1959[27]
	3.5 mg/l	Lepomis macrochirus	soft water; 30 C	Academy of Natural Sciences 1960[1]
	4.2 mg/l	"	soft water; 20 C	Academy of Natural Sciences 1960[1]
	12.5–12.9 mg/l	"	hard water; 20 & 30 C	Academy of Natural Sciences 1960[1]
	6.91 ppm	Lepomis macrochirus	continuous flow, acute bioassay; a,c,e,f; $ZnCl_2$; aerated distilled water	Cairns Jr. and Scheier unpublished 1955[142]
	20 ppm	"	BSA; a,c,e; $ZnCl_2$	Cairns Jr. 1957[25]
	0.6 ppm	Lepomis macrochirus fingerlings	zinc chlorate and sulfate used; 17.5 C, diluted, well water used; 4.5 ppm Ca	Lloyd 1960[75]
	4 ppm (48 hrs)	Lepomis macrochirus	LD50 value, BSA; a,c,d; zinc sulphate conc. as Zn	Herbert 1961[51]
	10 ppm	Limnodrilus hoffmeisteri	BSA; a,c,d,i; zinc sulfate	Wurtz and Bridges 1961[141]
	14 ppm	Physa heterostropha	same as above	"
	38.5 ppm	Asellus communis	same as above	"
	56 ppm	Argia sp.	same as above	"
	4.2 ppm (1 day)	Physa heterostropha	BSA; zinc sulphate	Wurtz 1962[140]
	1.9 ppm (2 day)	"	static acute bioassay, zinc sulfate	"
	1.9 ppm (3 day)	"	static acute bioassay zinc	"
	1.9 ppm	Physa heterostropha	same as above	"
	49.0 ppm (1 day)	Helisoma companulata	same as above	"
	49 ppm (2 day)	Helisoma companulata	static acute bioassay zinc	Wurtz 1962[140]
	13.4 ppm (3 & 4 day)	"	same as above	"
	10–12 ppm (48 hr)	Cyprinus carpio	pH 7.0–7.2; 28–30 C 8.8 mg/l CO	Sreenivasan and Raj 1963[120]
	10–15 ppm (48 hr)	Tilapia mossambica	"	"
	10 ppm (48 hr)	Danio sp	"	"
	3.86 ppm (2 day)	Salmo gairdneri	BSA; a,c,d,f; zinc sulphate	Herbert and Shurben 1964[53]
	26–40 ppm (48 hr)	Salmo gairdneri smolts	conc. as Zn using $ZnSO_4$; changing percent salinity; hardness 320 ppm; alk 240 ppm; aerated water	Herbert and Wakeford 1964[55]
	27 ppm–85 ppm (48 hr)	Salmo gairdneri	conc. as Zn. using $ZnSO_4$; hardness 320 ppm; alk. 240 ppm; aerated water; pH 7.8	"
	13.4 ppm	Helisoma campanulata	13 C; hard water; zinc sulfate (3.03 ppm Zn) time not given	Raymount and Shields 1964[105]
	3.85 ppm	"	13 C; in soft water $ZnSO_4$; 0.87 ppm Zn time not given	"
	0.04–2.00 ppm (1 day)	Salmo salar	continuous flow acute bioassay a,c,f; lab water had 3 μg/l Zn and 2 μg/l Cu	Schoenthal 1964[111]
	2.9–13.3 ppm	aquatic animals		Skidmore 1964[116]
	0.6 μg	Salmo salar	conc. as Zn, LC50 Value; Zn added as $ZnSO_4$ continuous flow bioassay; a,c,d,e,f	Sprague 1964[117]
	2.86–3.78 ppm	Lepomis macrochirus	18 C; in soft water; BSA; a,f	Cairns Jr. 1965[24]
	0.90–2.10 ppm	"	30 C; in soft water; BSA; a,f	"

TABLE 1—Continued

Constituent	Acute dose 96 hr LC50	Species	Conditions	Literature citation*
Zinc (Zn)	6.60–9.47 ppm	"	18 C; in hard water; BSA; a,f	"
	6.18–9.50 ppm	"	30 C; in hard water; BSA; a,f	"
	28 ppm (2 day)	Brachydanio rerio (adult)	BSA; a,c,d,e,f; distilled water, aerated; 24 C; 5–9 ppm DO; ZnCl$_2$; conc. as Zn	Cairns Jr. et al. 1965[29]
	105 ppm (2 day)	" (eggs)	same as above	"
	5.2 ppm (2 day)	Lepomis macrochirus	BSA; a,c,d,e,f; aerated distilled water; ZnCl$_2$; conc. as Zn; 24 C; 5–9 ppm DO	"
	3.9 ppm (2 day)	Salmo gairdneri	field study, river; a,c,e,f,l,m.	Herbert et al. 1965[57]
	0.96 mg l	Pimephales promelas	BSA; c,d,e,f; soft water zinc sulfate; conc. as Zn	Pickering and Henderson 1965[93]
	33.4 mg l	"	same as above using hard water	"
	5.46 mg l	Lepomis macrochirus	same as above using soft water	"
	40.9 mg l	Lepomis macrochirus	same as above using hard water	"
	6.44 mg l	Carassius carassius	same as above using soft water	"
	1.27 mg l	Lebistes reticulatus	same as above using hard water	"
	0.88 mg l	Pimephales promelas	BSA; c,d,e,f; zinc acetate; soft water; conc. as Zn	Pickering and Henderson 1965[93] 1966[94]
	5.37 mg l	Lepomis macrochirus	BSA; c,d,e,f; soft water; ZnCl$_2$; conc. as Zn	Pickering and Henderson 1965[93], 1966[94]
	1.69 mg l (12 days)	Pimephales promelas (eggs)		Pickering and Vigor 1965[95]
	3.95 ppm (1 day)	Pimephales promelas	BSA; a,c,d; zinc sulfate; tap water for eggs	"
	2.55 ppm (2 day)	"	same as above	"
	1.83 ppm	"	same as above	"
	1.71 ppm (7 day)	"	same as above	"
	1.63 ppm (12 day)	"	same as above	"
	0.95 ppm (1 day)	"	BSA; a,c,d; zinc sulfate; tap water, minnow fry	"
	0.95 ppm (2 day)	"	same as above	"
	0.87 ppm	"	same as above	"
	0.87 ppm (7 day)	"	same as above	"
	4.9 ppm	Pimephales promelas	continuous flow acute bioassay a,c,d,e; hardness 50 mg l; pH 8.0	Mount 1966[82]
	32.3 ppm	"	same as above with hardness 200 mg l and pH 6.0	"
Zinc (Zn)	4.6 ppm	Salmo gairdneri	static acute bioassay, c,e; zinc sulfate	Ball 1967[8]
	16.0 ppm (5 days)	Perca fluviatilis	same as above	"
	17.3 ppm (5 days)	Rutilus rutilus	same as above	"
	8.4 ppm (7 days)	Gobio gobio	same as above	"
	14.3 ppm (5 day)	Abramis brama	same as above	"
		juvenile salmon		Srpague and Ramsey 1965[119]
	5×10 [illegible] M to 7.5×10 [illegible] M	bryozoans, tubeworms, bivalve molluscs		Wisely and Blick 1967[137]
		Salmo gairdneri	alkylbenzene sulphonate used	Brown 1968[17]
	2.8–3.5 ppm	"	BSA; a,c,d,e,f,o	Brown et al. 1968[19]
	4.2 ppm	Lepomis macrochirus	BSA; a,c,d,e; Zn^{++}; all fish acclimatized for 2 weeks in syn. dil. water	Cairns Jr. and Scheier 1968[28]
	1 ppm (32 hrs)	Lebistes reticulatus	BSA; a,c,f,n,o	Chen and Selleck 1968[30]
	0.75 ppm (63 hrs)	Lebistes reticulatus	BSA; a,c,f,n,o	"
	0.56 ppm (96 hrs)	Lebistes reticulatus	BSA; a,c,f,n,o	"
	4.3 ppm (5 day)	Nitzschia linearis	BSA; a,c,e; ZnCl$_2$	Patrick et al. 1968[91]
	0.79–1.27 ppm	Physa heterostropha	BSA; a,c,e; ZnCl$_2$	Patrick et al. 1968[91]
	2.86–3.78 ppm	Lepomis macrochirus	BSA; a,c,e; ZnCl$_2$	"
	7.2 ppm (20 day)	Lepomis macrochirus	same as above; continuous flow acute bioassay. 1.8 mg l DO; a,c,d,e,f	Pickering 1968[92]
	12.0 ppm (20 day)	"	same as above with 5.6 mg l DO	"
	10.0 mg l (48 hr)	Penaeus duorarum	15 C; in the dark, ZnSO$_4$ conc. as Zn	Portmann 1968[97]
	100 mg l (48 hr)	Penaeus aztecus	"	"
	12 mg l (48 hr)	Hemigrapsis oregonensis	"	"
	200 mg l (48 hr)	Clinocardium nuttalli	"	"
	46.0 ppm	tubificid worm	static acute bioassay; a,c, zinc sulfate	Whitley 1968[135]
	7.5 ppm	Pimephales promelas		Rachlin and Perlmutter 1968[102]
	7.6 ppm	"	23 C (inbred strains)	"
	12.0 ppm	Xiphophorus	23 C "	"
	46 ppm (24 hr)	tubificid worm	pH 7.5	Whitley 1968[135]
	9.2 mg l promelas	Pimephales		Brungs 1969[20]
	10 ppm	Daphnia	TLm. Zn^{++}	Tabata 1969[121]

APPENDIX III–TABLE 2—Sublethal doses of inorganic chemicals for aquatic organisms

Constituent	Chronic dose	Species	Conditions	Literature Citation
Aluminum (Al)	106 mg/l	Daphnia magna	threshold of immobilization, in Lake Erie water; $Al_2(SO_4)_3$	Anderson 1944[149]
	190 ppm	Daphnia magna	threshold of immobilization, a,e, BSA; aluminum ammonium sulfate	Anderson 1944[149]
	206 ppm	Daphnia magna	threshold of immobilization; BSA, a,e; using aluminum potassium sulfate	"
	136 ppm	Daphnia magna	same, using aluminum sulfate	"
	<6.7 ppm	Daphnia magna	threshold of immobilization after 64 hrs; $AlCl_3$; BSA, a; 25 C	" 1948[151], 1950[152]
Ammonia (NH_3)	<134 ppm or 91 mg/l	Daphnia magna	threshold of immobilization in 64 hr, BSA, a,e, ammonium chloride, 25 C	Anderson 1944[149]
	<8.75 ppm	"	BSA; a,e; threshold of immobilization; ammonium hydroxide	"
	<106 ppm	"	threshold of immobilization; ammonium sulfate BSA; a,c	"
	8.75 mg/l	"	threshold conc. of immobilization using NH_4OH; 25 C	"
	17 ppm	Steurastrum paradoxum	inhibition of growth	Chu 1942[154]
	1000 ppm	Salmo gairdneri	loss of equilibrium in 27.3 min. in tap water; ammonium chloride; conc. as ammonia, a,c,e,f	Grindley 1946[155]
	1000 ppm	"	same as above, loss of equilibrium in 52.5 min.	"
	50 ppm	"	same as above; loss of equilibrium in >1000 min.	"
	3000 ppm	"	same as above using distilled water; loss of equilibrium in 292 min.	
	1000 ppm	"	same as above using distilled water; loss of equilibrium in 725 min.	"
	100 ppm	"	same as above using distilled water; loss of equilibrium in 4,320 mins.	"
	1000 ppm	Salmo gairdneri	loss of equilibrium in 29.8 min; tap water; BSA; a,c,e,f, ammonium sulfate; conc. as NH_3	Grindley 1946[155]
	3000 ppm	"	same as above using distilled water; loss of equilibrium in 318 mins.	"
	1000 ppm	"	same as above using distilled water; loss of equilibrium in 847 mins.	"
	100 ppm	"	same as above using distilled water; loss of equilibrium in >5,760 mins.	"
	91 ppm	Daphnia magna	BSA; a: threshold of immobilization in 64 hrs. ammonium chloride;	Anderson 1948[151]
	3.1 mg/l	Leptodora kindtii	threshold of immobilization	Anderson 1948[151]
	86 mg/l	Cyclops vernalis	"	"
	75 mg/l	Mesocyclops leukarti	threshold of immobilization	Anderson 1948[151]
Ammonia (NH_3)	5 mg/l	Diaptomus oregonensis	"	"
	152 mg/l	Daphnia magna	threshold of immobilization; using $(NH_4)_2SO_4$	"
	13 mg/l	Diaptomus oregonensis	same as above	"
	0.04 N	Gasterosteus aculeatus	immediate negative response	Jones 1948[205]
	0.01 N	"	reactions are slow; some are overcome by the exposure	"
	[illegible]1 mg/l	Daphnia magna	threshold of immobilization, 25 C; NH_4Cl	Anderson 1950[152]
	420 mg/l	Navicula seminulum	50 percent reduction of growth; soft water; 22 C	Academy of Natural Sciences 1960[146]
	420 mg/l	"	50 percent reduction of growth; hard water; 22 C	"
	320 mg/l	"	50 percent reduction of growth; soft water; 28 C	"
	420 mg/l	"	50 percent reduction of growth; hard water; 28 C	"
	410 mg/l	"	50 percent reduction of growth; soft water; 30 C	"
	350 mg/l	"	50 percent reduction of growth; hard water; 30 C	"
	420 mg/l	"	22 C in hard and soft water 50 percent reduction in division (growth)	"
	5.0–8.0 ppm (NH_3)	Oncorhynchus kisutch	in aerated fresh water, loss in equilibrium spasms with gills and jaws gaping	Holland et al. 1960[199]
	3.5–10.0 ppm	Oncorhynchus tshanytscha	in aerated salt water; reduction in growth; loss of equilibrium; Alk. 112 ppm; DO 8.4 ppm	" 1960[199]
Antimony (Sb) (See also Na)				
	37 ppm	Daphnia magna	threshold of immobilization; antimony trichloride; BSA; a	Anderson 1948[151]
	15 mg/l	protozoans	$K(SbO)C_4H_4O_6$; hindrance of food intake	Bringmann and Kuhn 1959[159]
	3.5 mg/l	green algae	" hindrance of cell division	"
	9 mg/l	Daphnia	" hindrance of movement	"
	1.0 mg/l	Micropterus salmoides	caused projectile vomiting $SbOH(C_4H_4O_6K_2)$ used	Jernejcic 1969[206]
Arsenic (As) (See also Sodium (Na) and Potassium (K))				
	20 ppm	Salmo gairdneri and minnows	conc. of arsenic using sodium arsenite, fish overturned in 36 hrs.	Grindley 1946[155]
	250 ppm		conc. of arsenic using sodium arsenate, fish overturned in 16 hrs.	"
	30–35 ppm	minnows	fins, scales damaged, diarrhea, heavy breathing and hemorrhage around fin areas.	Boschetti and McLoughlin 1957[152]
	4–10 µg	Mytilus edules	amount of As retained in flesh	Sautet et al. 1964[230]
	0.5–2 µg	Mytilus edules	amount of As retained in shell when exposed to 100 g/l of As	"

TABLE 2—Continued

Constituent	Chronic dose	Species	Conditions	Literature Citation
Arsenic (As)	100 g/l As	"	bysuss accumulated 250–500 µg	"
	100 g/l As	"	excreta contained 550–800 µg	"
	1.8 mg/l	Stizostedion vitreum vitreum (walleye)	as As (3.0 ml of arsenous acid) regurgitation of stomach contents into throat	Jernejcic 1969[204]
Barium (Ba)	< 83 ppm	Daphnia magna	threshold of immobilization, $BaCl_2$; BSA; a;c	Anderson 1944[149]
* (see also Sodium (Na) & Potassium (K))				
	12 mg/l	Leptodora kindtii	threshold of immobilization; 20–25 C; $BaCl_2$	Anderson 1948[151]
	133 mg/l	Cyclops vernalis	threshold of immobilization; 20–25 C; $BaCl_2$	Anderson 1948[151]
	5000 mg/l	fish	same as above	"
	29 ppm	Daphnia magna	BSA; a; threshold of immobilization; $BaCl_2$; 25 C	Anderson 1948,[151] 1950[152]
Beryllium (Be)	3 mg/l	Carassius auratus	using lagoon wastes from Be plant fish became sluggish after 21 days	Pomelee 1953[222]
	10^{-4}–10^{-2} M	Fundulus heteroclitus	conc. affecting liver enzyme activity	Jackim et al. 1970[211]
Boron	5000 mg/l	Salmo gairdneri	slight darkening of the skin using boric acid	Wurtz 1945[256]
	80,000 mg/l	"	caused immobilization and loss of equilibrium of fish; using boric acid	
	10 mg/l	marine fish	violent irritant response	Hiatt et al. 1953[197]
Bromine (Br) ** (see also Na)				
	< 0.0026 ppm	Daphnia magna	threshold of immobilization; $CaCl_2$; BSA; a; in 64 hrs.	Anderson 1948[151]
	10.0 ppm	marine fish	violent irritant activity caused by irritation of respiratory enzymes	Hiatt et al. 1953[197]
Cadmium (Cd)	0.0026 mg/l	Daphnia magna	threshold of immobilization	Anderson 1948[151], 1950[152]
	0.05–0.10 mg/l	Australorbis glabratus	produced distress syndromes; distilled water.	Harry and Aldrich 1958[illegible]
	142 ppm	Sewage organisms	50 percent inhibition of O_2 utilization; BOD; a; $CdSO_4$	Hermann 1959[195]
	0.1–0.2 ppm	Crassostrea virginica	20-week exposure; little shell growth lost pigmentation of mantle edge; coloration of digestive diverticulae	Shuster and Pringle 1969[236]
	50 ppm	Fundulus heteroclitus	pathological changes in intestinal tract, kidney, and gills; changes in essosinophil lineage	Gardner and Yevich 1970[illegible]
	10^{-4}–10^{-2} M	Fundulus heteroclitus	conc. affecting liver enzyme activity	Jackim et al. 1970[211]
Calcium (Ca)	1.332 ppm	Daphnia magna	threshold of immobilization; $CaCl_2$ BSA; a;c	Anderson 1944[149]
	920 ppm	"	threshold of immobilization; $CaCl_2$; BSA; a;c; 20–25 C	Anderson 1948[151]
	1730 mg/l	Cyclops vernalis	threshold of immobilization; 20–25 C using $CaCl_2$	Anderson 1948[151]
	1440 mg/l	Mesocyclops leukarti	same as above	"
	22,080 mg/l	white fish fry	same as above	"
	12,060 mg/l	pickerel fry	same as above	"
	8,400 ppm	Lepomis macrochirus	$CaCl_2$ 1.34 percent loss of tissue fluid; pH 8.3; 22.5 C; dissolution of mucous covering of body causing dehydration of musculature	Abegg 1950[143]
Calcium (Ca)	1.25×10^{-3} M	Cymnogaster aggregata	activation of brain acetylcholinesterase	Abou-Donia and Menzel 1967[144]
Chlorine (Cl)	0.3 ppm	trout	symptoms of restlessness, dysphea, loss of equilibrium & spastic convulsions	Cole 1941[167]
	10 mg/l (5 days)	Macrocystis pyrifera	10–15 percent reduction in photosynthesis	Clendenning and North 1960[165]
Chloride (see also sodium and potassium) (Cl^-)	2 mM	Salmo gairdneri	change in respiration rate	Amend et al. 1956[148]
Chromium (Cr) (see also sodium and potassium)				
	<0.6 ppm	Daphnia magna	chromic acid; threshold of immobilization; BSA a;c;	Anderson 1944[149]
	<3.6 ppm	"	threshold of immobilization; chromic chloride; BSA; a; 64 hrs.	" 1948[151]
	6.4–16.0 ppm	Chlorococcum variegatus	complete inhibition of growth for 56 days; Cr as dichromate.	Hervey 1949[196]
	3.2–6.4 ppm	Chlorococcum humicola	same as above	"
	3.2–6.4 ppm	Scenedesmus obliquus	same as above	"
	0.32–1.6 ppm	Lepocinclis steinii	same as above	"
	728 ppm	Lepomis macrochirus	hydration of tissues of body due to coagulation of mucous covering body; 22.5 C; pH 5.9	Abegg 1950[143]
	1.0 ppm	BOD	10 percent reduction in O_2 utilization; lab bioassay; j; chromic sulfate.	Ingols 1955[212]
	0.2 ppm	fish	retarded rate of growth and resulted in increased mortality (Cr^{++})	U.S. Dept of Commerce 1958[illegible]
	0.21 mg/l	Microregma	threshold effect	Bringmann and Kuhn 1959[159]
	not given	Salmo gairdneri	change in erythrocyte surface area and increase or decrease in haematocrit value	Harsband and Harsband 1963[illegible]
	2–4 mg/l	Salmo gairdneri	raising of hematocrits	Schiffman and Fromm 1959[231]
	2.5 ppm	"	Cr as chromate; lab bioassay; tap water; glucose transport by gut segments reduced 40 percent from controls.	Fromm and Stokes 1962[183]
	[illegible] ppm	fish	decreased extractable protein content of blended fish muscle	Castell et al. 1970[illegible]
Cobalt (Co)	>26 ppm	Daphnia magna	cobaltous chloride; BSA; a;c; threshold of immobilization.	Anderson 1944[149]
	>3.1 ppm	Daphnia magna	threshold of immobilization for 64 hr exposure BSA; a; $CoCl_2$	Anderson 1948[151]
	2.8 mg/l	Daphnia magna	threshold of immobilization using $CoCl_2$	Ohio River Valley Water Commission 1950[220]
	5 mg/l	Daphnia	threshold effects $CoCl_2$	Bringmann and Kuhn 1959[159]
	2.5 mg/l	E. coli	"	"
	1.0 mg/l	Scenedesmus	"	"
	0.5 mg/l	Microregma	"	"
	64.0 ppm	Sewage organisms	50 percent inhibition of O_2 utilization; BOD; a; $CoCl_2$	Hermann 1959[195]
	5 mg/l	Saphrolegnia	suppression of growth	Shabalina 1964[230]
	0.5, 0.05, and 5 mg/l	Cyprinus carpio (young)	inhibition of growth in small carp	"

TABLE 2—Continued

Constituent	Chronic dose	Species	Conditions	Literature Citation
Cobalt (Co)	5 mg/l	Saprolegnea	suppressed growth	"
	10–50 ppm	fish	decreased extractable protein content of blended fish muscle.	Castell et al. 1970
Copper (Cu)	1 µg/l Cu	Chlorella pyrenoidosa	suppressed growth; 4-hr exposure 20 C; 6µg/l Fe	Nielsen 1939
	2.5–.0 µg/l	Chlorella pyrenoidosa Nitzchia palea	decreased photosynthetic rate	"
	0.1 mg/l Cu	roach	cannot withstand conc. greater than given	Nielsen 1939
	2.0 ppm	large mouth black bass	in distilled water, CuSO lethal threshold	Cole 1941
	0.13 ppm	Crassostrea virginica	turn green in 21 days (unmarketable!)	Galtsoff 1943
	0.096 mg/l	Daphnia magna	threshold conc. of immobilization using cupric chloride	Anderson 1944
	0.1 ppm	Daphnia magna	threshold of immobilization using CuSO; BSA; a:c	Anderson 1944
	>0.2 mg/l	Bugula netritina	complete inhibition of growth of attached fauna	Miller 1946
	0.02–0.3 mg/l	barnacles	growth of young barnacles is inhibited	Miller 1946
	<0.2–0.3 mg/l	Bugula neritina	retarded growth	Miller 1946
	<0.2 mg/l		retarded polypide formation	
	0.027 ppm	Daphnia magna	threshold of immobilization, cupric chloride; BSA; a (64 hrs)	Anderson 1948
	2.7 mg/l	Cyclops vernalis	threshold of immobilization	Anderson 1948
	1.9 mg/l	Mesocyclops leukarti		"
	0.0024 mg/l	Diaptomus oregonensis		
	0.178 mg/l	Oncorhynchus gorbuscha	loss of equilibrium and initial mortalities (Cu NO)	
	metal sheet; 45 percent Ni 55 percent Cu	Balanus amphitrite	malformation of the shell bases; edges scalloped not smooth	Weis 1948
	0.027 mg/l	Daphnia magna	threshold conc. of immobilization using cupric chloride	Anderson 1950
	0.16 mg/l	sea urchin	as Cu; abnormalities occur in eggs	Cleland 1953
	0.1 mg/l	Australorbis glabratus	produced distress syndrome	Harry and Aldrich 1958
	21 ppm	Sewage organisms	50 percent inhibition of O utilization (BOD copper sulphate; a	Hermann 1959
	1.0 mg/l	Sphaerotilus	inhibition of growth conc. of CuSO	Academy of Natural Sciences 1960
	0.1–0.5 ppm	oyster	changes in digestive diverticulum tissues with desquamation and necrosis of stomach epithelium	Fujiya 1960
	0.563 ppm	Oncorhynchus gorbuscha (young)	loss of equilibrium and initial mortalities in 19 hrs; conc. as Cu; pH 7.9; Cu(NO)	Holland et al 1960
	1.00 ppm	Oncorhynchus kisuthh silver salmon	survival, growth, reproductive and feeding responses	Holland et al 1960
	16 ppm	Rana pipiens	chronic static bioassay; a;c; copper sulfate	Kaplan and Yoh 1961
	1.1 ppm	Salmo gairdneri	water, copper sulfate as Cu; BSA; a;e;p; 3 days; 3.5 ppm Zn	Lloyd 1961b
	0.044 ppm	Salmo gairdneri	same as above using 0.56 ppm Zn; and soft water (7 days)	Lloyd 1961b
	35–45 percent of incipient lethal level	Salmo salar	inhibition of migratory habits	Sprague and Saunders 1963
	0.1 mg/l	Nereis virens	threshold of toxicity; conc. as Cu. accumulation in gut and body wall.	Raymount and Shields 1964
	1–2 mg/l	Carcinus	threshold of toxicity; 11–12 day exposure	Reish 1964
	2.3 µg/l	Salmo salar	as Cu; threshold for avoidance for parr.	Sprague et al 1964
	0.42 µg/l	"	as Cu; plus 6.1 µg/l Zn; fish are 9.5–15.3 cm in length; avoidance	"
	0.7 ppm	goby	reduced appetite and reduced O consumption freshwater; pH 7.2; still water	Syazuki 1964
	1–5 ppm CuSO$_4$	Oncomelania formosana	decrease in food consumption; concentration of Cu along wall of digestive gland and in the loose spongy connective tissue of the stomach and proximal intestine	Winkler and Ch 1964
	10–20 µg/l	sea urchin	retards body growth of pluteal larvae, regressing of arms is retarded	Bougis 1965
	30 µg/l	sea urchin	affects growth of arms	"
	0.01–0.1 ppm	Helix pomatia	increase in mucous secretion and no response to tactile stimuli	DeClaventi 1965
	20 µg/l	oysters	green color in oysters	Sprague et al. 1965
	0.05 ppm	"	inhibition of self purification	"
	1.25 × 10 M	Cymatogaster aggregata	acetylcholinesterase activity is inhibited by Cu	Abou-Donia and Menzel 1967
	160 µg/l	common guppy	reduction in number of mucous cells	Cusick 1967
	0.06 ppm	Salmo salar	chronic static bioassay, CuSO as Cu	Grande 1967
	0.02 mg/l	Oncorhynchus	sublethal effects on fingerlings	Grande 1967
	< 1.0 ppm	crayfish Orconectes rusticus	inhibition of respiratory enzymes degenerative effect of cells and tissues including disruption of gluthathione equilibrium continuous flow bioassay	Hubschman 1967
	< 1.0 ppm	crayfish	same as above	"
	0.35–0.43 toxic units	Salmo salar	reduction in number of spawning salmon	Saunders and Sprague 1967
	0.056 ppm	Daphnia	inhibition of growth	Hueck and Adema 1968
	5.6 µg/l	Salmo gairdneri	threshold avoidance level	Sprague 1968
	0.055–0.265 µg/ml	dinoflagellates	growth inhibition at 20 C	Mandelli 1969
	0.025–0.05 ppm	oysters	bodies became bluish-green in color; and shell showed excellent growth; mantle edge pigmentation increased; and mortalities increased.	Shuster and Pringle 1969
	33 µg/l	Pimephales promelas	prevention of spawning hard water	Mount and Stephen 1969
	10–50 ppm	fish	decreased extractable protein content of blended fish muscle	Castell et al. 1970

TABLE 2—Continued

Constituent	Chronic dose	Species	Conditions	Literature Citation
Copper (Cu)	0.2 mg/l	Oncorhynchus tshawytscha	inhibition of growth	Hazel and Meith 1970[192]
	10^{-3}–10^{-4} M	Killifish	change in liver enzyme activity	Jackim et al. 1970
Cyanide (CN^-)	0.1–0.3 ppm	Crassius auratus	hard water using KCN; respiratory depressant	Cole 1941[167]
	0.126 mg/l	trout	overturned in 170 mins.	Ohio River Valley Water Commission 1950
	0.15 mg/l	trout	overturned in 170 mins. CN	Southgate 1950
	0.7 mg/l	Salmo gairdneri	fish overturned	Herbert and Merken 1952[193]
	1 ppm	fish	gills become brighter in colour due to inhibition by cyanide of the oxidase responsible for transfer of O_2 from blood to tissues	Southgate 1953
	5×10^{-3} M	Mayorella palestinensis	increased respiration of organism in glucose-containing solutions; a.c; BSA;	Reich 1955
	7300 mg/l	Chlorella	Inhibition of photosynthesis	Reich 1955
	0.1 mg/l	fish	fish overturned	Neil 1956
	1 ppm	fish	respiratory depressant—gills became brighter in color	Jones 1964
	0.25 ppm	goby, perch, mullet	change in O_2 uptake; reduction in appetite of some; still water; pH 8.2 KCN	Syazuki 1964
	10 mg/l	Lepomis macrochirus	3.0 mg/l free CO_2; conc. as CN; superficial coagulation of mucous; Alk 1.5 mg/l resulting in death of some; pH 6.0; CN complexed with silver;	Doudoroff et al. 1966
	not given	Cyprinus carpio minnow, gudgeon Rhodeus sericeus	loss of equilibrium, nervous system and respiration are effected	Malacea 1966
	2 mM CN	squid	affects the Ca efflux in the axons; after 90–150 min rate constant for loss of Ca was increased 5–10 fold.	Blaustein and Hodgkin 1969
Fluorine (F)	270 mg/l	Daphnia	23 C using NaF threshold effect	Bringmann and Kuhn 1959
	95 mg/l	Scenedesmus	24 C using NaF threshold effect	"
	226 mg/l	Microregma		"
	180 mg/l	Escherichia coli	27 C using NaF threshold effect	"
	500 ppm	Oncorhynchus kisutch	Alk 47.5 ppm; DO 8.4 ppm; after 72 hr exposure survivors were in poor condition, dark in color with light colored spots at end of snout	Holland et al. 1960[199]
	150 ppm	Salmo gairdneri	90 percent mortality in 21 days; BSA; a;dl; hard water	Herbert and Shurben 1964[194]
Iron (Fe)	2.0 mg/l	trout, salmon, roach	blockage of gills; Fe_2O_3	Nielson 1939[219]
	<152 ppm	Daphnia magna	BSA; a; c; thresholdl of immobilization $FeSO_4$	Anderson 1944[149]
	130 ppm	"	BSA; a; c; thresholdl of immobilization $FeCl_3$	Anderson 1944[149], 1950[152]
	<38 ppm	Daphnia magna	BSA; a; threshold of mobilization in 64 hrs;	Anderson 1948[151], 1950[152]
	5.0 ppm (1 day)	goby	reduction in appetite (in 1 day exposure using ferrous sulfate.	Syazuki 1964[249]
Iron (Fe)	27 μg/l	Phaeodactylim tricornutum	Severe clumping of diatom cells.	Davies 1966[172]
	1.25×10^{-3} M	Cymatogaster aggregata	inhibition of AChE activity	Abou-Donia and Menzell 1967[144]
	10–100 mg/l	Carassius auratus	epithelial edema, hypersecretion of mucous, inflammation, capillary congestion, destruction of respiratory epithelium, blockage of gill filaments and lamellae by microferruginous ppt. and occurrence of intra cellular iron in epithelial cells	Ashley 1970[153]
Lead (Pb)	5 mg/l	fish	precipitation of mucous of gills decreasing permeability of gills to dissolved O_2 (DO = 6.2 ppm)	Westfall 1945[254]
	1.25 ppm	Daphnia magna	threshold of immobilization: 64 hrs. $PbCl_2$ BSA; a	Anderson 1948[151]
	0.33–644 mg/l	tadpoles	negative reaction; lead nitrate	Jones 1948[205]
	0.04 N	Gasterosteus aculeatus	Fish reacted negatively then positively due to osmotic pressure of solution	Jones 1948[205]
	50 mg/l	catfish	injury to blood cells during exposure up to 183 days; conc. as lead acetate; in tap water	Doudoroff and Katz 1953
	30.6 ppm	barnacles	deformation of shells due to growth on unfavorable substrates.	Stubbings 1959[247]
	1.0 mg/l	Cyprinus carpio	harmed serum during long exposure; conc. as Pb	Fujiya 1961
	1.25 ppm	Poecilia reticulata	retardation of growth, increase in mortality, delayed sexual maturity	Crandall and Goodnight 1962[169]
	2.0 ppm	Lebistes reticulatus	chronic static bioassay $Pb(NO_3)_2$ retardation of growth, delay in sexual maturity and increased mortality 27 percent in 90 days.	Crandall and Goodnight 1962[169]
	1.25×10^{-4} M	Cymatogaster aggregata	inhibition of acetylchlonesterase activity	Abou-Donia and Menzel 1967[144]
	25 ppm	Rana pipiens	Sloughing of the skin after 20-days; loss of righting reflexes; loss of normal semi-erect posture	Kaplan et al. 1967[208]
	150 ppm	"	total loss of righting reflexes; excitement salivation, and muscular twitchings present upon 1st exposure; darkening of liver, gall bladder spleen & kidney observed	"
	1000 ppm	Rana pipiens	for 48 hrs. gastric mucosa eroded red blood cell and white blood cell counts decreased with increasing Pb.	Kaplan et al. 1967[208]
	10, 20, 40 mg/l	Lepomis cyanellus	avoided these concentrations	Summerfelt and Lewis 1967[248]
	25 mg/l	Salvelinus malma	reduction of growth	Dorfman and Whitworth 1969[175]

TABLE 2—Continued

Constituent	Chronic dose	Species	Conditions	Literature Citation
Lead (Pb)	10^{-7}–10^{-2} M	killifish	change in liver enzyme activity	Jackim et al. 1970[207]
	0.1–0.2 mg/l	Crassostrea virginica	induced changes in mantle & gonad tissue.	Pringle (unpublished)[258]
Magnesium (Mg)	50 ppm	Staurastrum paracloxum	certain inhibition of growth using $MgSO_4$	Chu 1942[164]
	740 ppm	Daphnia magna	BSA; a; threshold of immobilization $MgCl_2$	Anderson 1948[151]
	7.2 ppm	Botryococcus	inhibition of growth	"
Manganese (Mn) (see also Potassium (K) and Sodium (Na))				
	50 ppm	Daphnia magna	threshold of immobilization, $MnCl_2$ BSA; a	Anderson 1948[151], 1950[152]
	50 mg/l	Daphnia magna	as Mn, threshold of immobilization; 23 C	Bringmann and Kuhn 1959[159]
	1.25×10^{-4}	Cymatogaster aggregata	activation of acetylcholinesterase	Abou-Donia and Menzel 1967[144]
	10,000 ppm	Lebistes reticulatus	inhibition of essential sulfhydryl groups attached to key enzyme, lab bioassay	Shaw and Grushkin 1967[231]
	10,000 ppm	Bufo valliceps	same as above; using tadpoles	"
	1,000 ppm	Daphnia magna	as above	"
Mercury (Hg)	<0.006 ppm	Daphnia magna	threshold of immobilization; $HgCl_2$; a; BSA;	Anderson 1948[151]
	0.61 ppm	Sewage organisms	50 percent inhibition of O_2 utilization; $HgCl_2$ BOD; a;	Hermann 1959[195]
	3.2×10^{-6} mg/hr	Japanese eel, Crassius auratus	accumulation in the kidney	Hibiya and Oguri 1961[197]
	0.01 ppm	Lebistes reticulatus	cation combined with essential sulfhydryl group attached to a key enzyme to cause inhibition; a;c;e; BSA	Shaw and Grushkin 1967[231]
	0.1 ppm	Bufo valliceps	same as above using tadpoles	
	0.1 ppm	Daphnia magna	same as above	"
	10^{-7}–10^{-2} M	killifish	change in liver enzyme activity	Jackim et al. 1970[207]
Molybdenum (Mo)	54 mg/l	Scenedesmus	threshold conc. for deleterious effect	Bringmann and Kuhn 1959[159]
Nickel (Ni)	<0.7 ppm	Daphnia magna	threshold of immobilization $Ni(NH_4)_2(SO_4)_2$ BSA; a;	Anderson 1948[151]
	0.7 mg/l	Daphnia	threshold of immobilization; $NiCl_2$	Anderson 1950[152]
	1.5 mg/l	Scenedesmus	threshold of immobilization; $NiCl_2$	Bringmann and Kuhn 1959[159]
	0.1 mg/l	E. coli	threshold of immobilization; $NiCl_2$	"
	0.05 mg/l	Microregma	threshold of immobilization; $NiCl_2$	"
	1.25×10^{-4} M	Cymatogaster aggregata	inhibition of acetylcholinesterase activity	Abou-Donia and Menzel 1967[144]
	10 ppm	Lebistes reticulatus	bioassay; a;c;e; cation combined with essential sulfhydryl group attached to key enzyme to cause inhibition.	Shaw and Grushkin 1967[231]
	100 ppm	Bufo valliceps	same as above, using tadpoles	"
	10 ppm	Daphnia magna	same as above.	"
	0.5–10 mg/l	Cyanophyta	growth inhibition	Sparling 1968[241]
Nitrate	0.0007 N	minnow	as $Pb(NO_3)_2$; showed negative response	Jones 1948[205]
	1.25×10^{-1} M	Cymatogaster aggregata	$Pb(NO_3)_2$; caused 73 percent inhibition of AChE activity	Abou-Donia and Menzel 1967[144]
	10, 20–40 mg/l	Lepomis cyanellus	avoided these concentrations	Summerfelt and Lewis 1967[245]
pH	pH 9.0	oyster larvae	injury to larvae	Gaardner 1932[184]
	pH 4.0	fish	coagulation of proteins of epithelial cells	Cole 1941[167]
pH	62 ppm	Daphnia magna	threshold of immobilization; HCl BSA; a;c	Anderson 1944[149]
	pH 2.8	Crassius auratus	coagulation of mucous on gills; H_2SO_4	Westfall 1945[254]
	pH 5.4	Gasterosteus aculeatus	reacted negatively to pH less than 5.4 and greater than 11.4	Jones 1948[205]
	pH 11.4	"	"	"
	pH 6.5	oyster	pumping is reduced	Korringa 1952[209]
	pH 5.51 (3 day)	Oncorhynchus tshawytscha	0.1 N HCl; critical level, flowing-salt	Holland et al. 1960[199]
	50–150 ppm	short-necked clam	O_2 uptake became abnormal; increase in consumption with 24-hr. exposure.	Syazuki 1964[239]
Potassium (K)			inhibition of growth	Chu 1942[164]
	0.6 ppm	Daphnia magna	threshold of immobilization; $K_2Cr_2O_7$; BSA; a;c	Anderson 1944[149]
	0.63 ppm	Daphnia magna	threshold of immobilization; BSA; a;c; $KMnO_4$	Anderson 1944[149]
	373 ppm	Daphnia magna	threshold of immobilization; BSA; a;c; KCl	Anderson 1944[149]
		"	loss of equilibrium in 23.8 mins. $K_2Cr_2O_7$; BSA; a;c;e;f	Grindley 1946[188]
	1000 ppm	Salmo gairdneri	loss of equilibrium in 54.6 mins.; BSA; $K_2Cr_2O_7$ tap water; conc. as Cr	"
	200 ppm	Salmo gairdneri	loss of equilibrium in 188 min. BSA; $K_2Cr_2O_7$ tap water conc. as Cr	Grindley 1946[188]
	2000 ppm	Salmo gairdneri	loss of equilibrium; in 42.0 mins; K_2CrO_4; BSA; a;c;e;f; conc. as Cr	Grindley1946[188]
	1000 ppm	"	loss of equilibrium in 79 mins. BSA; K_2CrO_4; conc. as Cr; a;c;e;f	"
	20 ppm	"	loss of equilibrium in 3580 min; BSA; K_2CrO_4 conc. as Cr; a;c;e;f	"
	432 ppm	Daphnia magna	threshold of immobilization; BSA; a;c; KCl for 64 hrs	Anderson 1948[151]
	10.5 ppm	Sewage organisms	50 percent reduction of BOD values; K_2CrO_4	Sheets 1957[233]
	15 ppm	sewage organisms	50 percent inhibition of O_2 utilization; BOD; KCN; a	Hermann 1959[195]
	17.0 ppm	sewage organisms	50 percent inhibition of O_2 utilization; BOD; a $K_2Cr_2O_7$	Hermann 1959[195]
	0.072 ppm	Rabora heteromorpha	20 percent mortality in 7 days; KCN; BSA	Abram 1964[143]
Selenium (Se)	>800 ppm	fresh-water fish	accumulation of Se in liver, from bottom deposits in reservoir	Barnhart 1958[154]
	2.5 mg/l of Se	Daphnia	medium threshold effect using sodium selenite; 23 C	Bringmann and Kuhn 1959[159]
	2.5 mg/l of Se	Scenedesmus	median threshold level; using sodium selenite; 24 C	"
	90 mg/l of Se	Escherichia coli	median threshold level, using sodium selenite; 27 C	"
	183 mg/l of Se	Microregma	median threshold level, using sodium selenite	"
Silver (Ag)	6×10^{-6}–3.3×10^{-4} M	Bacterium coli	inhibits enzymes; 20 C Ag_2SO_4	Yudkin 1937[257]
	0.0051 ppm	Daphnia magna	threshold of immobilization; BSA; (64 hrs) silver nitrate; a;	Anderson 1948[151]

TABLE 2—Continued

Constituent	Chronic dose	Species	Conditions	Literature Citation
Silver (Ag)	0.03 mg/l	Daphnia	median threshold effect	Bringmann and Kuhn 1959[150]
	0.03 mg/l	Microregma	"	"
	0.05 mg/l	Scenedesmus	"	"
	0.04 mg/l	Eschericha coli	"	"
	0.15 μg/l	Echinid larvae		Soyer 1963[240]
	10–100 μg/l	Paracentrotus	as $AgNO_3$; abnormalities or inhibition of growth of eggs	"
	2μg/l	"	as $AgNO_3$; delay in development and deformation of resulting pluter	"
	0.25 μg/l	"	threshold conc. for effect; as $AgNO_3$	"
	0.50 μg/l	Arbacia	threshold conc. for effect for eggs	"
	0.1 ppm	Lebistes reticulatus	cation combines with essential sulfhydryl group attached to key enzyme causing inhibition; BSA; a;c;e	Shaw and Grushkin 1967
	0.1 ppm	Bufo valliceps	same as above using tadpoles	"
	0.1 ppm	Daphnia magna	same as above	"
	10^{-1}–10^{-2} M	Fundulus heteroclitus	change in liver enzyme activity	Jackim et al. 1970
Sodium (Na)	6143 ppm NaCl	Daphnia magna	threshold of immobilization; BSA; NaCl; a;c	Anderson 1944[149]
	8500 ppm	Daphnia magna	BSA; a;c; threshold of immobilization; $NaNO_3$	Anderson 1944[149]
	<3.4 ppm	Daphnia magna	threshold of immobilization; BSA; NaCN	Anderson 1946[150]
	5000 ppm	"	threshold for immobilization; unfavorable osmotic effect exerted; BSA; $NaNO_3$	Anderson 1946
	9.4 ppm	"	conc. causing immobilization; BSA; Na_2S	Anderson 1946
	<0.32 ppm	Daphnia magna	threshold of immobilization; BSA; Na_2CrO_4	Anderson 1946
	8200 ppm	"	same as above using NaBr	Anderson 1946
	210 ppm	"	threshold of immobilization; BSA; $NaBrO_3$	Anderson 1946
	9.1 ppm	Daphnia magna	threshold of immobilization; BSA; $NaAsO_2$	Anderson 1946
	953 ppm	Phoxinus phoxinus	loss of equilibrium in 54.6 min; BSA; a;c; e;f; $NaAsO_2$; tap or dist. water; conc. as As	Grindley 1946
	250 ppm	"	loss of equilibrium in 186 min; BSA; a;c;e; f; $NaAsO_2$; tap water or dist. water	"
	17.8 ppm	"	loss of equilibrium in 2174 mins; BSA; a;c;e;f; $NaAsO_2$; tap or dist. water	"
	<20 ppm	Daphnia magna	threshold of immobilization; BSA; sodium arsenate	Anderson 1946
	2970 ppm	Phoxinus phoxinus	lost equilibrium in 205 mins; BSA; a;c;e;f; dist. or tap water; sodium arsenate dist. or tap water	Grindley 1946
	820 ppm	"	lost equilibrium in 467 mins; BSA; sodium arsenate; a;c;e;f; dist. or tap water	"
	234 ppm	"	lost equilibrium; a;c;e;f; dist or tap water; 951 min sodium arsenate	"
Sodium (Na)	3680 ppm	Daphnia magna	threshold of immobilization; NaCl BSA a;c;	Anderson 1948[151]
	0.007 N	Gasterosteus aculeatus	fish displayed distress; tap water; BSA; c;e; pH 6.8 with H_2SO_4; Na_2S	Jones 1948[205]
	2.47 ppm	Daphnia magna	50 percent are immobilized in 100 hr exposure; BSA; a;c; Na_2SiO_3	Freeman and Fowler 1953[179]
	158 ppm	"	50 percent are immobilized in 100 hr exposure; BSA; a;c; Na_2SiO_3; plus 2.899 ppm Na_2SO_4	"
	0.0003 N	Gasterosteus aculeatus	survival time of 72 hrs. tap water. BSA; c;e; pH 6.8; Na_2S	Jones 1948[205]
	0.201 ppm	Daphnia magna	50 percent immobilization; BSA; 100 hr exposure a;c; Na_2CrO_4; plus 119 ppm Na_2SiO_3 & 2180 ppm Na_2SO_4	Freeman and Fowler 1953[179]
	0.276 ppm	"	50 percent immobilization during 100 hr exposure BSA; a;c; Na_2CrO_4; plus 2984 ppm Na_2SO_4	"
	0.150 ppm	"	50 percent immobilization for 100 hr exposure; BSA; a;c; Na_2CrO_4; plus 93 ppm Na_2SiO_3	"
	0.33 ppm	"	50 percent immobilization during 100 hr exposure Na_2CrO_4 plus 408 ppm Na_2CO_3; BSA; a;c;	Freeman and Fowler 1953[179]
	85 ppm	"	50 percent immobilization, 100-hr exposure; a;c; BSA; Na_2SiO_3 plus 180 ppm Na_2SO_4	"
	86 ppm	"	50 percent immobilization; 100 hr exposure; a;c; BSA; Na_2SiO_3 plus 182 ppm Na_2CO_3 plus 0.146 ppm Na_2CrO_4	"
	0.195 ppm	Daphnia magna	50 percent immobilization; 100 hr exposure; a;c; BSA; Na_2CrO_4 plus 240 ppm Na_2CO_3 and 2079 Na_2SO_4	"
	73 ppm	"	50 percent immobilization, 100 hr exposure; a;c; BSA; Na_2SiO_3 plus 155 ppm Na_2CO_3 and 1343 ppm Na_2SO_4	"
	0.35 ppm	"	50 percent immobilization; Na_2CrO_4; BSA; a;c; 100 hr exposure; plus 87 ppm sodium bisulfate and 440 ppm sodium carbonate conc. as Na_2CrO_4	"
	92 ppm	"	50 percent immobilization; Na_2SiO_3; BSA; a;c; 100-hr exposure; plus 38 ppm $NaHSO_3$; and 194 ppm Na_2CO_3	"
	427 ppm	Daphnia magna	50 percent immobilization; Na_2SiO_3 BSA; a;c; 100 hr exposure; plus 177 ppm $NaHSO_3$	Freeman and Fowler 1953[179]
	0.286 ppm	"	50 percent immobilization; Na_2CrO_4; BSA; a;c; 100 hr exposure; 70 ppm $NaHSO_3$	"

TABLE 2—Continued

Constituent	Chronic dose	Species	Conditions	Literature Citation
Sodium (Na)	126 ppm	"	50 percent immobilization Na_2SiO_3; BSA; a;c; 100 hr exposure; + 52 ppm $NaHSO_3$ and 2308 ppm Na_2SO_4	"
	506 ppm	"	50 percent immobilization Na_2SiO_3; BSA; a;c; 100 hr exposure; plus 144 ppm $NaHSO_3$ and 0.861 ppm Na_2CrO_4	"
	0.306 ppm	"	50 percent immobilization Na_2CrO_4; BSA; a;c; 100 hr exposure; plus 75 ppm $NaHSO_3$ and 3312 ppm Na_2SO_4	"
	0.42 ppm	"	50 percent immobilization; BSA; 100 hr exposure; a;c; Na_2CrO_4	"
	1.0 ppm	sewage organisms	j; 100 percent reduction in O_2 utilization BOD; Na_2CrO_4	Ingols 1955
	3.6 ppm	sewage organisms	reduction by 50 percent in the BOD values; BOD; NaCN	Sheets 1957
	100 ppm	sewage organisms	50 percent inhibition of O_2 utilization; BOD; a; sodium arsenate	Hermann 1959
	4 ppm	Cladophora	complete decomposition in 2 weeks; field study in lake; a;c; $NaAsO_2$	Cowell 1965
	4 ppm	Spirogyra zygnema	same as above	Cowell 1965
	4 ppm	Potamogeton (plant)	same as above	"
	4 ppm	zooplankton	$NaAsO_2$; field study in lake; a;c; significant reduction evident	"
	6.5 ppm	Daphnia magna	median immobilization concentration; a;c;d; e;g; BSA; $NaAsO_2$	Crosby and Tucker 1966
	1.4 ppm	Simocephalus serrulatus	threshold of immobilization; $NaAsO_2$; BSA; 78 F	Sanders and Cope 1966
	1.8 ppm	Daphnia magna	same as above	"
Sulfide ($S^=$)	5.0 ppm	suckers	causes respiratory paralysis	Cole 1941
	0.86 ppm	sunfish	"	"
	3.8 ppm	Salvelinus malma	"	"
	4.3 ppm	Crassius auratus	"	"
	6.3 ppm	Cyprinus carpio	"	"
	3.2 mg/l	trout	overturned in 2 hrs.; pH 9.0	Southgate 1948
	3.2 mg/l	trout	overturned in 10 mins.; pH 7.8	"
	3.2 mg/l	trout	overturned in 4 mins.; pH 6.0	"
Titanium (Ti)	4.6 mg/l of Ti	Daphnia	median threshold level; 23 C	Bringmann and Kuhn 1959
	2.0 mg/l of Ti	Scenedesmus	median threshold effect; 24 C	"
	4.0 mg/l of Ti	Microregma	median threshold level;	"
Uranium (U)	13 mg/l	Daphnia	threshold effect of uranyl nitrate; as U	Bringmann and Kuhn 1959
	22 mg/l	Scenedesmus	threshold effect of uranyl nitrate; as U	"
	1.7–2.2 mg/l	Escherichia coli	threshold effect of uranyl nitrate; as U	"
	28 mg/l of U	Microregma	threshold effect of uranyl nitrate	"
	0.5 mg/l of U	Escherichia coli	disturbs O_2 balance of water and inhibits development of enteric bacteria	Guskova and Griffein 1964
Zinc (Zn)	0.1 mg/l	roach	cannot withstand	Nielson 1939
	48 ppm	Daphnia magna	threshold of immobilization; BSA; a;c; zinc sulfate	Anderson 1944
Zinc (Zn)	25 ppm	Salmo gairdneri	loss of equilibrium in 133 min; a;c;e;f; zinc sulfate; conc. as Zn; BSA;	Grindley 1946
	24 mg/l	fish	avoidance concentration of $ZnSO_4 \cdot 7H_2O$	Jones 1948
	0.15 mg/l of Zn	Daphnia magna	threshold conc. of zinc immobilization using $Zn(NO_3)_2$	Anderson 1950
	0.04 mg/l of Zn	rainbow trout	prevention of hatching of rainbow trout eggs in soft water.	Affleck 1952
	0.16 mg/l	Psammechinus miliaris	abnormalities of fertilization cleavage of eggs of urchins when in zinc sulfate; conc. of Zn.	Cleland 1953
	1 mg/l	Planorbis and Bulinus (snails)		Deschiens et al. 1957
	920 ppm	sewage organisms	reduction in BOD values by 50 percent zinc sulfate	Sheets 1957
	56 ppm	sewage organisms	reduction of BOD value by 50 percent in an unbuffered system; zinc borofluoride.	Sheets 1957
	0.75 ppm	sewage organisms	reduction of BOD value by 50 percent in an unbuffered system; zinc cyanide	Sheets 1957
	1.8 mg/l	Daphnia magna	median threshold effect; as Zn	Bringmann and Kuhn 1959
	1.4–2.3 mg/l	Escherichia coli	same as above	"
	1.0–1.4 mg/l	Scenedesmus	same as above	"
	0.33 mg/l	Microregma	same as above	"
	1.25 ppm & 230 ppm	Poecilia reticulata (common guppy)	retardation of growth, increased maturity and delayed sexual maturity; as Zn; $ZnSO_4$	Crandall and Goodnight 1962
	35–45 percent of incipient lethal level.	Salmo salar	migration of salmon is disturbed when copper-zinc pollution exceeds this dosage	Sprague and Saunders 1963
	100 mg/l	lobster	causes increase in Zn levels in urine, excretory organs, hepatopancreas and gills	Bryan 1964
	0.0–5.0 ppm	Lepomis macrochirus Lepisosteus osseus Dorosoma petenense Dorosoma cepedianum Alosa chrysochloris Cyprinus carpio Carassius auratus	continuous flow bioassay, acute; a;c;f; accumulation of Zn in bones and gills.	Mount 1964
	53.3 mg/l	Salmo salar	avoidance response in 50 percent of fish; BSA; a;c;d;e;f; conc. as Zn	Sprague 1964
	53 mg/l	Salmo salar	avoidance conc. for parr; conc. as Zn.	Sprague et al. 1964
	12.6 ppm	shellfish	decrease in O_2 uptake in presence of Zn sulfate as Zn; 1 hr exposure in polluted sea water.	Syazuki 1964
	30 ppm	goby	rate of O_2 uptake is decreased, reduction of appetite; as zinc; 1 day exposure	Syazuki 1964
	0.15 ppm	oysters	green color evident; cause inhibition of self-purification	"

TABLE 2—Continued

Constituent	Chronic dose	Species	Conditions	Literature Citation
Zinc (Zn)	160 µg/l	Poecilia reticulata	zinc damaged epithelium of gills, reduction in the number of mucous cells; pH 6; distilled water, high mortality rate	Cusick 1967[171]
	157 & 180 ppm	Fundulus heteroclitus	as Zn; sluggish and uncoordinated after 2 hrs; DO. 7.2–7.4 ppm; 20 C; pH 8.0; salinity 25 ‰	Eisler 1967[176]
	10.0 ppm	Lebistes reticulatus	bioassay; a;e;c; combines with essential sulfhydryl group attached to a key enzyme.	Shaw and Grushkin 1967[236]
	10.0 ppm	Bufo valliceps	same as above (using tadpoles)	"
	1.0 ppm	Daphnia magna	same as above	"
	0.35–0.43 toxic units	Salmo salar	reduction in number of salmon reaching spawning grounds (avoidance reactions of migrating salmon)	Saunders and Sprague 1967[229]
Zinc (Zn)	0.8 mg/l	Salmo gairdneri	histological damage to gills; Zn added along with alkylbenzene sulfonate	Brown et al. 1968[160]
	100 µg/l	freshwater mussels	accumulation of Zn in Leydig cells and mucous cells of the epithelial layers	Pauley and Nakatani 1968[221]
	5.6 µg/l	Cyanophyta	avoidance reactions to sub-lethal conc. of Zn, low avoidance threshold	Sprague 1968[243]
	5.6 µg/l	Salmo gairdneri	avoidance reactions	Sprague 1968[243]
	0.18 mg/l	Pimphales promelas	reproduction inhibited; no effect on survival growth or maturation.	Brungs 1969[161]
	18.0 ppm	Salmo gairdneri	reduction of mitotic index of gonadal cells by 70 percent	Rachlin and Perlmutter 1969[223]
	32.0 ppm	Salmo gairdneri	complete inhibition of mitotic division	"
	16 ppm (24 hr)	Cyprinus carpio	hardness 25 ppm Ca; as Zn	Tabata 1969[251]

APPENDIX III-TABLE 3—Accumulation of inorganic chemicals for aquatic organisms

Constituent	Concentration in sea water	Species	Tissue or organ	Concentration in tissue	Concentration factor	Literature Citation
Barium(Ba)	....	Gracilaria foliifera	....	....	1,200–13,000	Bedrosinn 1962[263]
Cadmium (Cd)	^{115}Cd 12 µc/l+2 mg/l stable Cd	Chasmychthys gulosus	viscera	....	3.6 (6 days)	Hiyama and Shimizu 1964[280]
	″	″	dig. tract	....	15 (3 days)	″
		″	gill	....	3.0 (2 days)	″
		″	skin	....	0.3 (2 days)	″
		″	scales	....	2.2 (10 days)	″
		″	vertebrae	....	0.18 (3 days)	″
		″	muscle	....	0.077 (3 days)	″
		″	head and fins	....	0.37 (8 days)	″
	^{115}Cd 12 µc/l+20 µg/l stable Cd	Ulva pertusa	whole	....	11 (4 days)	″
		Venerupis philippinarum	mantle gill	....	58 (8 days)	″
		....	....	....	>100 (3 days)	″
	″	″	adductor	....	8.3 (3 days)	″
	″	″	other viscera	....	52 (8 days)	″
	″	″	shell	....	>3	″
	″	Leander sp.	viscera	....	>250	″
	″	″	muscle	....	0.38 (1 day)	″
	″	″	shell	....	725	″
	^{115}Cd 12 µc/l+20 µg/l stable Cd	Strongylocentrotus pulcherrimus	digestive tract	....	110 (1.5 days)	Hiyama and Shimizu 1964[280]
	″	″	gonad	....	>8	″
	″	″	aristotle's lantern	....	>3	″
	″	″	test	....	>10	″
		Chasmychthys gulosus	viscera	....	>10	″
	″	″	digestive tract	....	>6	″
	″	″	gill	....	11 (6 days)	″
	″	″	skin	....	0.92 (6 days)	″
	″	″	scales	....	0.80 (5 days)	″
	″	″	vertebrae	....	0.22 (3 days)	″
	″	″	muscles	....	0.16 (4 days)	″
	″	″	head and fins	....	0.96 (9 days)	″
	″	Venerupis philippinarum	gill	....	19 (1 day)	″
	″	″	mantle	....	9.8 (1.5 days)	″
	″	″	adductor	....	5.1 (3 days)	″
	″	″	other viscera	....	8.3 (1.5 days)	″
	″	″	shell	....	>1	″
	16 mg/l (5 days)	Lepomis macrochirus	gill	634 µg/kg	....	Mount and Stephan 1967[294]
	8 mg/l (30 days)	″	″	252 µg/kg	....	″
	20 mg/l (20 days)	″	″	484 µg/kg	....	″
Calcium (Ca)	38 mg/100 ml CaCl$_2$	Daphnids	fresh weight after 48 hrs.	138.3 mg/100 g	....	Korpincnikov et. al. 1956[289]
	946 µc/201 22 C	Tilapia mossambica	fish tissue	2.7×10^{-7} µc/gm	Ca. 0.6	Boroughs et. al. 1957[264]
	not measured	Lebistes	spine	....	62. ±0.4	Rosenthal 1957[300]
	8.52×10^{3} cpm/ml	″	body	....	0.72±0.01	″
	9.42×10^{4} cpm/ml	Lebistes	body	....	0.72±0.003 (10 days)	Rosenthal 1957[300]
	7.37×10^{5} cpm/ml	″	″	....	0.82±0.004 ″	″
	not measured	″	Carcass	....	1.00±0.045 ″	″
	″	″	head	....	1.07±0.039 ″	″
	″	″	viscera	....	0.59±0.087 ″	″
	″	″	muscle	....	0.102±0.024 ″	″
	″	″	spine	....	1.87±0.10 ″	″
	″	″	carcass	....	100.0±2.92	″
	″	″	head	....	21.3±1.12	″
	″	″	viscera	....	7.3±0.48	″
	″	″	muscle	....	3.7±0.31	″
	1 µc Ca^{45}Cl$_2$	Fucus vesiculosus	thallus	90 percent uptake in 24 hours	....	Swift and Taylor 1960[305]
		Ceramium rubrum	″	....	100–300	Taylor and Odum 1960[306]
		Enteromorpha intestinalis	″	....	100–300	″
	7.37×10^{5} cpm/ml	Lebistes (15 days)	Whole	5.5×10^{6} cpm/10 mg	....	Rosenthal 1963[301]
	10^{5} cpm/ml	″	spine (10 days)	2.8×10^{5} cpm/100 mg	....	″
		″	head (10 days)	1.7×10^{5} cpm/100 mg	....	″
		″	total (10 days)	1.4×10^{5} cpm/100 mg	....	″
		″	viscera (10 days)	1.2×10^{5} cpm/100 mg	....	″
		″	muscle (10 days)	$.2\times10^{5}$ cpm/100 mg	....	″
	10^{5} cpm/ml	Danio	whole (22 days)	2.8×10^{5} cpm/100 mg	....	″
Chromium (Cr)	10–13.0 µc injected into air bladder	Crassius auratus	intestine	25 cpm/mg	....	Hibiya and Oguri 1961[278]
		″	liver	25–40 cpm/mg	....	″
		″	pancreas	25–40 cpm/mg	....	″
		″	spleen	60–100 cpm/mg	....	″
		″	kidney	200 cpm/mg	....	″
		″	head Kidney	275 cpm/mg	....	″
		″	gill	40–60 cpm/mg	....	″
		″	muscle	10 cpm/mg	....	″
		″	backbone	30–40 cpm/mg	....	″

TABLE 3—Continued

Constituent	Concentration in sea water	Species	Tissue or organ	Concentration in tissue	Concentration factor	Literature Citation
Chromium (Cr)		Crassius auratus	gonad	30-60 cpm/mg		Hibiya and Oguri 1961[273]
		"	air bladder	1.000 cpm/mg		"
	0.204 pCi/ml	Lampsilis radiata	soft tissues	89.6 pCi/g	440	Harvey 1969[277]
Chromium (Cr^{+3})	17,804 cpm/g	Hermione	whole	10,373 cpm/g (9 day)	0.59	Chipman 1967[268]
	17,833 cpm/g	"	"	5,410 cpm/g (11 day)	0.31	"
	18,226 cpm/g	"	"	3,713 cpm/g (22 day)	0.21	"
	0.31 µg/l	"	"		3.0 (5 days)	"
		"	"		3.5 (7.5 days)	"
		"	"		5.0 (9.0 days)	"
		"	"		7.5 (12.5 days)	"
		"	"		8.0 (15.0 days)	"
		"	"		12.0 (19.0 days)	"
	0.1 µg/l	"	"	8.1 µg/g (1 day) (dry)		"
	0.3 µg/l	"	"	0.4 µg/g (2 day) (live)		"
	0.3 µg/l	"	"	0.7 µg/g (3 day) (live)		"
	"	"	"	0.9 µg/g (5 day) (live)		"
	"	"	"	1.1 µg/g (7 day) (live)		"
	0.3 µg/l	"	"	1.3 µg/g (9 day) (live)		Chipman 1967[268]
	"	"	"	1.7 µg/g (12 day) (live)		"
	"	"	"	2.3 µg/g (15 day) (live)		"
	"	"	"	2.7 µg/g (19 day) (live)		"
	3.0 µg/l	"	"	14.0 µg/g (4 day) (live)		"
	"	"	"	22.0 µg/g (8 day) (live)		"
	"	"	"	26.0 µg/g (11 day) (live)		"
	"	"	"	34.0 µg/g (15 day) (live)		"
	10 µg/l	"	"	24 µg/g (2 day)		"
	"	"	"	40 " (4 day)		"
	"	"	"	53 " (6 day)		"
	"	"	"	68 " (8 day)		"
	"	"	"	84 " (11 day)		"
	10 µg/l	Hermione	whole	106 µg/g (13 day)		Chipman 1967[266]
	100 µg/l	"	"	206 " (3 day)		"
	"	"	"	288 " (6 day)		"
	"	"	"	428 " (11 day)		"
	"	"	"	495 " (14 day)		"
	500 "	"	"	856 " (3 day)		"
	"	"	"	1139 " (6 day)		"
	"	"	"	1436 " (11 day)		"
	"	"	"	1834 " (14 day)		"
Chromium (Cr)	1 = conc. of phytoplankton culture—Cr transferred down food chain	Mummichog	gonad		9.0	Baptist and Lewis 1967[261]
		"	muscle		0.5	"
		"	gills		1.7	"
		"	spleen		6.9	"
		"	liver		1.7	"
		"	dig. tract		2.2	"
	(132 MCi/mg = initial conc. in phytoplankton culture.)	Zooplankton, post-larvae fish	whole		9.9	"
			"		73.	"
			"		6.2	"
	1 µCi CrCl$_3$/l	Podophthalmus vigil	gills	5000 dpm/mg(max) (2 days)		Sather 1967[304]
	"	"	muscle	79-80 dpm/mg (") (2-4 days)		"
	"	"	midgut gland	75 dpm/mg (max) (6 days)		"
	"	"	carapace	50 dpm/mg(max) (14 days)		"
		"	blood	10 dpm/mg(max) (16 days)		"
	5.3 µCi 51 CrCl$_3$ injected	"	gills	3000 (max) (16 days)		"
		"	midgut glands	1000 (max) (5 days)		"
				800 dpm/mg(max) (0-8 days)		"
Cobalt (Co)		Gadus macrocephalus			36	Ichikawa 1961[304]
		Chelidonichthys kumu			82	"
		Evynnis japonica			20	"
		Lateolabrax japonicus			30	"
		Seriola quinqueradiata			14	"
		Germo germo	whole		28	"
		Katsuworms vagans	"		84	"
		Scomber japonicus	"		28	"
		Cololabis saira	"		84	"
		Sardinops melanosticta	"		64	"
		Cleipea pallasii	"		26	"
		Stichopus tremulus	"		240	"
		Palinurus sp.	"		4,000	"
		Polypus sp.	"		52	"
		Ommastrephes sloani	"		62	"
		Ostrea gigas	"		170	"
		Pecten yessoensis	"		190	"
		Meretrix meretrix lusoria	"		200	"
		Porphora sp.	"		64	"

TABLE 3—Continued

Constituent	Concentration in sea water	Species	Tissue or organ	Concentration in tissue	Concentration factor	Literature Citation
Cobalt (Co)		Laminaria sp.	whole		27	Ichikawa 1961[284]
		Monostroma sp.	"		15	"
	2 µg l	Chasmichthys gulosus	"		0.101 (0.25 days)	Hiyama and Khan 1964[279]
	"	Chasmichthys gulosus	"		0.511 (1 day)	"
	"	Chasmichthys gulosus	"		1.57 (2 days)	"
	"	Chasmichthys gulosus	"		2.89 (4 days)	"
	"	Chasmichthys gulosus	"		4.58 (6 days)	"
	"	Chasmichthys gulosus	"		4.56 (9 days)	"
	2.54 dpm ml	Cambarus longulus longerostris 0.60g	whole	166 dpm animal	164	Wiser and Nelson 1964[310]
	25.4 dpm ml	" 0.49g	"	1,071 dpm animal	90	"
	254 dpm ml	" 0.54g	"	8,984 dpm animal	81	"
	2540 dpm ml	Cambarus longulus longerostris 0.45g	"	46,000 dpm animal	63	"
	25,400 dpm ml	" 0.65g	"	785,000 dpm animal	68	"
	254,000 dpm ml	" 0.55g	"	8,761,000 dpm animal	61	"
	2.54 dpm ml	" 0.60g	"	793 dpm animal	624	"
	25.4 dpm ml	" 0.80g	"	3921 dpm animal	213	"
	254 dpm ml	" 0.54g	"	23,322 dpm animal	145	"
	2540 dpm ml	" 0.45g	"	78,881 dpm animal	216	"
	25,400 dpm m	" 0.55g	"	35,874,000 dpm animal	334	"
	254,000 dpm m	" 0.43g	"	26,900,000 dpm animal	203	"
	6.81 × 10^5 dpm animal (average)	Cambarus longulus longerostris	gut	1.12 × 10^5 dpm g		Wiser and Nelson 1964[310]
		"	blood	2.93 × 10^3 dpm g		"
		"	muscle	3.09 × 10^4 dpm g		"
	"	"	gonad	2.97 × 10^4 dpm g		"
	"	"	integument	2.28 × 10^5 dpm g		"
	"	"	hepatopanoreas	1.96 × 10^5 dpm g		"
	Black Sea	Ulva rigida	whole		335	Polikarpov et al. 1967[296]
	N. W. Pacific	Ulva persuda	"		380	"
	Black Sea	Cystoseira barbata	"		45	"
	N. W. Pacific	Sargassum thumbergii	"		420	"
	Black Sea	Leander adspersus	"		11	"
	N. W. Pacific	Leander pacificus	"		7	"
	0.0006–0.015 ppm	Crassostrea viginica	flesh	99.3–1153 ppm	0.6 × 10^5	Preston 1967[297]
	0.0008–.0240 ppm	Crassostrea viginica	flesh	313–3174 ppm	2.4 × 10^5	Preston 1967[297]
	0.0023–0.0026	"	"	361–863 ppm	2.5 × 10^5	"
	0.027 p Ci ml	Lampsiles radiata	soft tissues	21.3 p Ci g	790	Harvey 1969[277]
	Alakanuk	Oncorhunchus tshawytscha (King salmon)	muscle F. & M.		9,400	Jenkins 1969[285]
	Alaska		liver		50,000	"
	"	Oncorhynchus keta	roe		42,000	"
	"	Chum salmon	muscle		13,000	"
	"		liver		32,000	"
	"	"	roe		60,000	"
	Kenai, Alaska	Oncorhynchus nerka	muscle M.		6,000	"
	"	(Sockeye Salmon)	muscle F.		3,200	"
		Salmon	liver		22,000	"
		Oncorhynchus nerka	roe		28,000	"
		Salmon	bone		11,000	"
	Seward	Oncorhynchus kisutch (silver salmon)	muscle F.		6,400	"
		"	muscle M.		7,200	"
		"	livers		33,000	"
		"	roe		37,000	"
	4.5 × 10^{-5} µCi/125 ml	Plectonema boryanum	whole cell	0.36 µCi/g (7 days)	6,200 (25 C)	Harvey 1969[277]
		"	"	0.32 µCi/g "	4,500 (30 C)	"
		"	"	0.28 µCi/g "	3,500 (35 C)	"
	4.5 × 10^{-5} µCi/125 ml	Plectonema boryanum	whole cell	0.18 µCi/g (7 days)	2,500 (40 C)	Harvey 1969[277]
	^{60}Co pCi l	Tridacna crocea	kidney	56000 pCi/g		Welander 1969[306]
		Plankton	whole	100 pCi/g		"
		Sea invertebrates	whole	950 pCi/g		"
		Fish	whole	18 pCi/g		"
		Algae	whole	8.8 pCi/g		"
	0.47 pCi/l ^{57}Co	Plankton	whole	15.0 pCi/g		Welander 1969[306]
		Sea Invertebrates	whole	11.0 pCi/g		"
		Fish	whole	0.69 pCi/g		"
	1.2 pCi/l	Plankton	whole	58 pCi/g		Welander 1969[306]
	^{60}Co	Algae	whole	33 "		"
		Sea invertebrates	kidney	2600 "		"
		Fish	liver	130 "		"
Copper (Cu)	0.002 N sol'n	Fundulus heteroclitus	dried flesh	0.0100 percent (1 hr)		White and Thomas 1912[309]
	"	"	"	0.0164 percent (3 hrs.)		"
	"	"	"	0.0230 percent (4 hrs.)		"
	"	"	undried flesh	0.00226 percent (1 hr)		"
	"	"	"	0.00360 percent (3 hrs)		"
	"	"	"	0.00529 percent (4 hrs)		"

TABLE 3—Continued

Constituent	Concentration in sea water	Species	Tissue or organ	Concentration in tissue	Concentration factor	Literature Citation
Copper (Cu)	N/1000 sol'n. $CuSO_4$	Tautoga onitis	whole (dry)	0.008 percent		"
			blood system	0.010 percent Cu Dry		"
	"	"	alimentary tract	0.003 percent "		"
	"	"	residue	0.005 percent "		"
	"	"	flesh	0.009 percent "		"
				percent by weight of Cu in dried flesh		
	.004 N CuSo sol'n	Fundulus heteroclitus	dried flesh	0.0160 percent (1 hr)		"
		"	"	0.0156 percent (2 hr)		"
	"	"	"	0.0[illegible]01 percent (3 hr)		"
	0.7 μg/l	Lampsilis radiata	soft tissues	1.6 μg/g	228.5	Harvey 1969[271]
Gold (Au)	oral dose	blue crab	gills	.7 percent of oral dose after 4 days		Duke et al. 1966[272]
	"	"	muscle	.6 percent of oral dose after 4 days		"
	"	"	carapace	.08 percent of oral dose after 4 days		"
		"	blood	.04 percent of oral dose after 4 days		"
	"	croaker	kidney	0.01 percent of oral dose after 148 hours		"
	"	"	gills	.056 percent of oral dose after 148 hours		"
	"	croaker	skin (scales)	.009 percent of oral dose after 148 hours		Duke et al. 1966[272]
	"	"	liver	.02 percent of oral dose after 148 hours		"
	"	"	muscle	.03 percent of oral dose after 148 hours		"
	"	"	heart	.0008 percent of oral dose after 148 hours		"
	"	"	spleen	.042 percent of oral dose after 148 hours		"
	"	"	gonad	.001 percent of oral dose after 148 hours		"
	"	blue crab	dig gland	12 percent of oral dose after 4 days		"
	"	"	stomach-gut	3 percent of oral dose after 4 days		"
	"	"	gonads	.8 percent of oral dose after 4 days		"
Iron (Fe)	$1.1\times10^{-}$ mg/ml	Dactylopterus volitans (Gurnard)	flesh	$0.9\times10^{-}$ mg/g of fish		Aten et al. 1961[260]
		Mackerel	flesh	$1.0\times10^{-}$ "		"
		Melanogrammus aeglefinus (Haddock)	flesh	$6\times10^{-}$ "		"
		Whiting	flesh	$0.4\times10^{-}$ "		"
		Plaice	flesh	$2\times10^{-}$ "		"
		Cod	flesh	1.2×1.0 mg/g "		"
	0.01 mg/l	Trachurus japonicus	whole		700	Ichikawa 1961[284]
	"	Pleuronectes sp.	"		600	"
	"	Scomber japonicus	"		1,800	"
	"	Cololabis saira	"		3,000	"
	"	Lateolabrax japonicus	"		3,000	"
	"	Chrysophyrus major	"		400	"
	"	Sardinops maleanostricta	"		2,000	"
	"	Theragra chalcogramma	"		400	Ichikawa 1961[284]
	"	Clupea pallasii	"		1,800	"
	"	Acanthogobius flarimanus	"		2,000	"
	"	Anthocidaris crassispina	"		10,000	Ichikawa 1961[28]
	"	Stichopus japonicus	intestine		78,000	"
	"	Panulirus lobster	"		1,000	"
	"	Penaeus (common shrimp)	"		1,000	"
	"	Paneaeopsis sp. shrimp	"		4,000	"
	"	Paralithodes camtschatica			4,000	"
	"	Neptunus marine crab			2,000	"
	"	Octopus fangsiao			6,000	"
	"	Turbo cornutus			9,000	"
	0.01 mg/l	Haliotus gigantea			3,000	"
	"	Haliotus diversicolor			17,000	"
	"	Meretrix meretrix lusoria			13,000	"
		Venerupis japonica	Whole		7,000	"
		Ostrea gigas	"		8,000	"
		Porphyra tenera	"		2,000	"
		Gelidium amansii	"		4,000	"

TABLE 3—Continued

Constituent	Concentration in sea water	Species	Tissue or organ	Concentratoin in tissue	Concentration factor	Literature Citation
Iron (Fe)		Laminaria sp	whole		5,800	Ichikawa 1961[294]
		Undario pinnatifida	"		1,300	"
		Hizikia fusiforme	"		2,900	"
	0.00004 nCi/kg	Phytoplankton	"	0.5 n Ci/kg		Palmer and Beasley 1967[295]
	0.00004 n Ci/kg	Euphausids	"	1.5 n Ci/kg		Palmer and Beasley 1967[295]
	"	Mytilus	"	0.36 n Ci/kg		"
	"	kelp	"	0.03 "		"
	"	Lepas (barnacle)	"	140.0 "		"
	"	squid	muscle	0.76 "		"
	"	squid	liver	8.6 "		"
	"	purple sea cucumber	whole	76.0 "		"
	"	sea urchins	whole	0.48 "		"
	Black sea	Ulva rigida	whole		730	Polikarpov et al. 1967[296]
	N. W. Pacific	Ulva persuda	whole		100	"
	750 cpm/g	quahog	shell	9.5×10^3 cpm/g		Andrews and Warren 1969[259]
			tissue	670 cpm/g		"
			feces	1.2×10^7 cpm/g		"
	380 cpm/g	clam	shell	1.1×10^3		"
			tissue	2.3×10^4		"
			feces	1.7×10^6		"
	4.5×10^{-5} µCi/125 ml	Plectonema boryanum	whole cell	0.18 µCi/g (2 day)	2,600 (25 C)	Harvey 1969[277]
			"	0.17 µCi/g (2 day)	2,400 (30 C)	"
	"	"	"	0.21 µCi/g (2 day)	2,700 (35 C)	"
	"	"	"	0.23 µCi/g (2 day)	3,200 (40 C)	"
	3.3 mg/l Fe	Mytilus edulis	soft tissue		5.5	Hobden 1969[291]
	3.4 days 1.0 mg/l	"	"	1.5 µg/g	1.5	"
	1.0 mg/l	Mytilus edulis L.	digestive gland		5. (3–4 day) (max)	Hobden 1969[291]
	3.3 mg/l	"	"		5.4 (2–3 day) (max)	"
	1.0 mg/l	"	gills		1.3 (average)	"
	3.3 mg/l	"	gills		1.8 (max) (0.3 day)	"
	1.0 mg/l	"	mantle		1.0 (max) (1 day)	"
	3.3 mg/l	"	mantle		.4 (max) (2.7 day)	"
	124 pCi/l of ^{55}Fe	Algae, fish	whole	480 p Ci/g		Welander 1969[308]
	"	"	muscle	80 pCi/g		"
	"	"	liver	264,000 p Ci/g		"
Manganese (Mn)		Clupea harengus	whole		95	Ichikawa 1961[294]
		Gadus sp.	"		320	"
		Scomber sp.	"		80	"
		Pleuronectes sp.	"		70	"
		Stichopus regalis	"		200	"
		Sepia officinalis	"		10,000	"
		Octopus vulgaris	"		50,000	"
		Haliotus tubercalata	"		750	"
		Pectan jacobaeus	"		10,000	"
		Ostrea edulis	"		1,500	"
		Mactra corallina	"		620	"
		Ulva lactuca	"		1,300	"
		Enteromorpha sp.	"		1,500	"
		Laminaria saccharina	"		300	"
		Fucus serratus	"		7,500	"
	1000 µg/l for 15 days, animals were starved	Homarus vulgaris " 200-350 g	whole blood	3.9 µg/g wet tissue 15 day		Bryan and Ward 1965[267]
			abdominal muscle	0.8 "		"
			hepalopancreas	4.1 µg/g wet tissue 15 day		"
			gills	26.9 "		"
			shell	207 "		"
			teeth of gastric mill	106 "		"
			stomach fluids	1.6 "		"
			hind gut and rectum	3.4 "		"
			excretory organs	5.1 "		"
			ovary	3.3 "		"
		Homarus vulgaris 200–350 g.	whole blood	2.4 µg/g wet tissue 15 day		Bryan and Ward 1965[267]
			abdominal muscle	0.8 "		"
			hepatopancreas	4.8 "		"
			gills	20.8 "		"
			shell	225 "		"
		Homarus vulgaris 200–350 g.	teeth of gastric mill	155 µg/g wet tissue		Bryan and Ward 1965[267]
	1000 µg/l Mn	Homarus vulgaris	Carapace edge	236 µg/g		Bryan and Ward 1965[267]
		Homarus vulgaris	whole blood		1.42	Bryan and Ward 1965[267]
	absorption in 72 hours	" (728 g)	urine		0.66	"
	"	"	stomach fluid		0.18	"
	"	"	abdominal muscles		0.15	"
	"	"	hepatopancreas		2.30	"
	"	"	gills		2.14	"
	"	"	excretory organs		1.91	"
	absorption in 72 hours	Homarus vulgaris	ossicles and teeth		1.42	"

TABLE 3—Continued

Constituent	Concentration in sea water	Species	Tissue or organ	Concentration in tissue	Concentration factor	Literature Citation
Manganese (Mn)	2.0 µc/l absorption in 72 hours	Homarus vulgaris (728 g)	shell carapace	96.7 mµc/g	7.06	Bryan and Ward 1965[267]
	"	"	shell claw	18.7 "	1.37	"
	"	"	shell telson	181.0 "	13.2	"
	"	"	whole animal	52.5 "	3.82	"
	2.0 µc/l in sea water plus 10 mg Mn in stomach absorption in 72 hrs.	Homarus vulgaris (744 g)	whole blood	27.4 "	2.20	"
		"	urine	12.3 "	0.99	"
		"	stomach fluid	3.8 "	0.31	"
	"	"	abdominal muscle	2.1 "	0.17	"
	"	"	hepatopancreas	6.1 "	0.49	"
	"	"	gills	36.7 "	2.96	"
	"	"	excretory organs	17.3 "	1.40	"
	"	"	ossicles and teeth	31.8 "	2.56	"
	"	"	shell, carapace	134.0 "	10.8	"
	"	"	shell, claw	85.6 "	6.91	"
	"	"	shell, telson	151.0 "	12.2	"
	2.0 µc/l in sea water plus 10 mg Mn in stomach absorption in 72 hrs.	Homarus vulgaris (744 g)	whole animal	59.5 mµc/g	4.80	Bryan and Ward 1965[267]
	2 µg/l normal sea water specimens; unstarved	Homarus vulgaris 200–350 g.	excretory organs	3.7 µg/g		Bryan and Ward 1965[267]
		"	ovary	1.6 "		"
	10 mg Mn pipetted into stomach	Homarus vulgaris 320 g.	blood	(7 hr) 65 µg/g		"
	"	"	hepatopancreas	(2 ") 165 "		"
	"	"	stomach fluid	(2 hr) 385 "		"
	"	"	urine	(7 ") 85 "		"
	"	"	muscle	(7 ") 10 "		"
	"	"	shell	(7 ") 205 "		"
	"	"	ossicles and teeth	(2 ") 130 "		"
	"	"	excretory organs	(7 ") 100 "		"
	"	"	gills	(7 hr) 55 "		"
	0.3 µCi/l Mn^{54}	Anodonta nuttalliana	calcareous tissue	97,000 cpm/g		Harrison 1967[276]
		"	mantle	45,000 cpm/g		"
	"	"	gills	29,000 cpm/gm		"
	0.3 µCi/l Mn^{54} + 0.1 ppm Mn	Anodonta nuttalliana	adductor muscle	14,000 cpm/g		Harrison 1967[276]
	"	"	dig. gland and stomach	18,000 cpm/g		"
	"	"	gonad and intestine	11,000 cpm/g		"
	"	"	body fluid	3,000 cpm/g		"
	0.033 ppm stable Mn	Unio	shell	762 ± 9.5 µg/g	2.3$\times 10^{4}$	Merlini 1967[293]
	"	5.1–6.0 cm	gills	14185 ± 12 "	6.0$\times 10^{4}$	"
	"	"	mantle	13088 ± 1470 "	5.5$\times 10^{4}$	"
	"	"	visceral sac	3571 ± 835 "	1.5$\times 10^{4}$	"
	"	"	adductor muscle	2539 ± 411 "	1.1	"
	0.033 ppm stable Mn	Unio 6.1–7.0 cm	shell	892 ± 13.0 "	2.6$\times 10^{4}$	"
	"	"	gills	18257 ± 1179 "	7.7$\times 10^{4}$	"
	0.033 ppm	Unio 6.1–7.0 cm	mantle	17765 ± 581 µg/g	7.5$\times 10^{4}$	Merlini 1967[293]
	"	"	visceral sac	4308 ± 307 "	1.8$\times 10^{4}$	"
	"	"	adductor muscle	2565 ± 296 "	1.1$\times 10^{4}$	"
	"	Unio 7.1–8.0 cm	shell	956 ± 21.0 "	2.8$\times 10^{4}$	"
	"	"	gills	20737 ± 1972 "	8.8$\times 10^{4}$	"
	"	"	mantle	19659 ± 984 "	8.3$\times 10^{4}$	"
	"	"	visceral sac	5034 ± 622 "	2.1$\times 10^{4}$	"
	"	"	adductor muscle	3067 ± 319 "	1.3$\times 10^{4}$	"
	0.0004 pCi/ml ^{54}Mn	Unio 5.1–6.0 cm	shell		0.82$\times 10^{4}$	"
		"	gill		3.6$\times 10^{4}$	"
		"	mantle		3.0$\times 10^{4}$	"
	0.2 ppm stable Mn	Unio 7.1–8.0 cm	visceral sac	5070 ± 1095 µg/g	7.6$\times 10^{4}$	"
		"	adductor muscle	2514 ± 504 "	1.8$\times 10^{4}$	"
	0.00015 pCi/ml ^{54}Mn	Unio 4.1–5.0 cm	shell		1.9$\times 10^{4}$	"
		"	gill		43.0$\times 10^{3}$	"
	0.00015 pCi/ml Mn^{54}	Unio 4.1–5.0 cm	mantle		22.0$\times 10^{3}$	Merlini 1967[293]
		"	visceral sac		5.6$\times 10^{3}$	"
	"	Unio 5.1–6.0 cm	shell		1.6$\times 10^{4}$	"
	"	"	gill		35.0$\times 10^{3}$	"
	"	"	mantle		26.0$\times 10^{3}$	"
	"	"	visceral sac		9.5$\times 10^{3}$	"
	"	Unio 6.1–7.0 cm	shell		2.0$\times 10^{4}$	"
	"	"	gill		30.0$\times 10^{4}$	"
	"	"	mantle		28.0$\times 10^{4}$	"
	"	"	visceral sac		8.6$\times 10^{4}$	"
	"	Unio 7.1–8.0 cm	shell		1.8$\times 10^{4}$	"
	"	"	gill		30.0$\times 10^{4}$	"
	"	"	mantle		23.0$\times 10^{4}$	"
	"	"	visceral sac		7.6$\times 10^{3}$	"
	0.004 pCi/ml Mn^{54}	Unio 5.1–6.0 cm	visceral sac		1.1$\times 10^{4}$	"
		Unio 6.1–7.0 cm	shell		0.68$\times 10^{4}$	"
	"	"	gills		3.1$\times 10^{4}$	"
	"	"	mantle		3.7$\times 10^{4}$	"

TABLE 3—Continued

Constituent	Concentration in sea water	Species	Tissue or organ	Concentration in tissue	Concentration factor	Literature Citation
Manganese (Mn)	0.004 pCi/ml	Unio 6.1–7.0 cm	visceral sac		1.1×10^{3}	Merlini 1967[293]
	Mn	Unio 7.1–8.0 cm	shell		0.53×10^{4}	"
	"	"	gills		3.8×10^{3}	"
	"	"	mantle		4.0×10^{3}	"
	"	"	visceral sac		1.1×10^{4}	"
	0.02 ppm stable Mn	Unio 5.1–6.0 cm	shell	299 ± 64.0 µg/g	1.5×10^{3}	"
	"	"	gills	1254 ± 1292 "	8.7×10^{4}	"
	"	"	mantle	7576 ± 986 "	5.3×10^{3}	"
	"	"	visceral sac	2154 ± 212 "	1.5×10^{4}	"
	"	"	adductor muscle	2008 ± 29 "	1.4×10^{4}	"
	"	Unio 4.1–6.1 cm	shell	225 ± 5.6 "	1.1×10^{3}	"
	"	"	gills	11391 ± 649 "	8.0×10^{3}	"
	"	"	mantle	4805 ± 489 "	3.4×10^{3}	"
	"	"	adductor muscle	1104 ± 115 "	0.77×10^{3}	"
	0.02 ppm stable Mn	Unio 6.1–7.1 cm	shell	378 ± 26.0 µg/g	1.9×10^{4}	Merlini 1967[290]
	"	"	gills	18154 ± 1562 "	13.0×10^{3}	"
	"	"	mantle	15008 ± 1288 "	10.0×10^{4}	"
	"	"	visceral sac	4964 ± 553 "	3.5×10^{3}	"
	"	"	adductor muscle	2055 ± 135 "	1.4×10^{3}	"
	"	Unio 7.1–8.0 cm	shell	515 ± 31.5 µg/g	2.5×10^{3}	"
	"	"	gill	20279 ± 616 "	14.0×10^{3}	"
	"		mantle	16316 ± 703 "	11.0×10^{4}	"
	Calif.	Mytilus edulis	whole		830	Polikarpov et al. 1967[296]
	Calif.	Mytilus californicus	whole		800–830	"
	1.4 µg/l	Laminaria digitata	plant	0.33 µg/g	236	Bryan 1969[266]
	4.5×10[illegible] µCi/125 ml	Plectonema boryanum	whole cell	0.13 µCi/g	15,350 (25 C)	Harvey 1969[277]
		"	"	0.16 "	27,700 (30 C)	"
	"	"	"	0.19 "	35,300 (35 C)	"
	"	"		0.17 "	27,900 (40 C)	"
	0.013 pCi/ml	Lampsiles radiata	soft tissue	30.9 pCi/g	2380	Harvey 1969[277]
	"	"	clam shell	15.0 "	1150	"
Mercury (Hg)	0.2 mg/l using HgCl$_2$	Elminius	whole body	0.92 mg/l dry wt.		Corner and Rigler 1958[274]
	1000 mg/l using HgCl$_2$	Artemia	whole body	0.47 mg/l "		"
	50 mg/l Hg using HgCl$_2$	Leander serratus	Branchiostegite	4.3 mg/g dry wt.		"
	"	"	Pleopods	0.48 mg/g dry wt.		"
	"	"	dorsal chitin	0.13 "		"
	"	"	gills	0.49 mg/g "		"
	"	"	antennary gland	0.32 mg/g "		"
	"	"	hepatopancreas	0.02 mg/g "		"
	"	"	central nervous system	0.04 mg/g "		"
	"	"	muscle	0.00 mg/g "		"
	10 µg/l Hg injected in 0.01 ml sea water as HgCl$_2$	Leander serratus	branchiostegite	13.0 µg/g dry wt.		Corner and Rigler 1958[274]
		"	pleopods	2.2 "		"
		"	dorsal chitin	3.4 "		"
		"	gills	29.3 "		"
	10 µg/l Hg injected in 0.01 ml sea water as HgCl$_2$	"	antennary gland	13.3 µg/g dry wt.		"
		"	hepatopancreas	4.4 "		"
		"	central nervous system	3.5 "		"
	"	"	muscle	2.7 "		"
	7.6 µc injected dose into air bladder	Crassius auratus	intestine	1000 cpm/mg		Hibiya and Oguri 1961[278]
		"	liver	1000 "		"
	"	"	pancreas	1500 "		"
	"	"	spleen	1500 "		"
	"	"	kidney	11000 "		"
	"	"	head kidney	2500–3000 "		"
	"	"	gill	200–300 "		"
	"	"	muscle	100–200 "		"
	"	"	backbone	100–200 "		"
	"	"	gonad	400–700 "		"
	"	"	air bladder	900–1400 "		"
	0.06 ng/g Hg using mercuric nitrate	Cod	blood	2.511 ng/g (7 days)	39.2	Hannerz 1968[275]
		"	heart	4.574 "	71.47	"
	"	"	liver	0.876 "	13.69	"
	"	"	spleen	1.998 "	31.22	"
	"	"	gonads	0.4412 "	6.89	"
	"	"	kidneys	1.529 "	23.89	"
	"	"	stomach	1.248 "	19.50	"
	"	"	brains	0.190 "	2.97	"
	"	"	eyes	0.270 "	4.22	"
	"	"	gills	234.784 "	3668.20	"
	"	"	fins	7.173 "	112.08	"
	"	"	scales	5.620 "	87.81	"
	"	"	muscles	0.21162 "	3.38	"
	"	"	bones	0.675 5 days	10.55	"
	"	"	heart	1.711 "	19.72	"

TABLE 3—Continued

Constituent	Concentration in sea water	Species	Tissue or organ	Concentration in tissue	Concentration factor	Literature Citation
Mercury (Hg)	0.06 ng/g Hg using mercuric nitrate	Cod	liver	0.365 "	5.70	Hannerz 1968[275]
	"	"	spleen	0.913 "	14.27	"
	"	"	gonads	0.487 "	7.61	"
	"	"	kidneys	0.798 "	12.47	"
	"	"	stomach	0.670 "	10.47	"
	"	"	brains	(0.193) "	3 92	"
	"	"	eyes	0.153 "	2.39	"
	"	"	gills	147 818 "	2309 70	"
	"	"	fins	3.443 "	53.79	"
	"	"	scales	3.865 "	60.39	"
	"	"	muscles	0.105 "	1.64	"
	"	"	bones	0.250 "	3.91	"
	0.05 ng/g Hg using mercuric chloride (mean value)	Glossosiphonia complanata	whole	...	670 (65 days)	"
	"	Herpobdella octoculata	"	...	534 "	"
	"	sludge worms	"	...	517 "	"
	"	Planorbis sp.	"	...	414 "	"
	"	Lynmaea stagnalis	"	...	293 "	"
	"	Physa fontinalis	"	...	637 (14 days)	"
	"	Ephemeroptera larvae	"	...	138 (65 days)	"
	"	"	"	...	28 (14 days)	"
	"	Trichoptera larvae	"	...	513 (49 days)	"
	"	Tipula	"	...	517 "	"
	"	Chironomidae larvae	"	...	175 "	"
	"	"	"	...	362 (65 days)	"
	0.05 ng/g Hg using $HgCl_2$ (mean value)	damselfly nymphs	"	...	655 "	"
	"	Hydrophilidae larvae	"	...	603 "	"
	"	Corixa sp.	"	...	414 "	"
	"	Notonecta glauca	"	...	483 "	"
	"	Gerris	"	...	431 "	"
	"	Planorbis sp.	"	...	560 (one month)	"
	"	Lymnaea stagnalis	"	...	247 "	"
	"	Corixa sp.	"	...	431 "	"
	0.30 ng/g Hg mercuric chloride	Pike	blood	176 ng/g (8 days)	587	Hannerz 1968[275]
	"	"	heart	258 "	860	"
	"	"	liver	377 "	1.258	"
	0.30 ng/g Hg mercuric chloride	Pike	spleen	608 ng/g (8 days)	2.027	Hannerz 1968[275]
	"	"	gut	199 "	663	"
	"	"	kidneys	495 "	1.653	"
	"	"	gonads	107 "	357	"
	"	"	eyes	36 "	120	"
	"	"	brain	284 "	947	"
	"	"	gills	878 "	2.928	"
	"	"	scales	214 "	713	"
	"	"	fins	406 "	1.353	"
	"	"	muscles	26 "	87	"
	"	"	bone	56 "	187	"
	0.06 ng/g Hg mercuric nitrate	Cod	blood	0.29 ng/g (2 days)	4.8	"
	"	"	heart	0.58 "	9.7	"
	"	"	liver	0.08 "	1.3	"
	"	"	spleen	0.35 "	5.8	"
	"	"	kidneys	0.21 "	3.5	"
	0.06 ng/g Hg mercuric nitrate	Cod	gut	0.20 ng/g (2 days)	3.3	Hannerz 1968[275]
	"	"	brain	0.15 "	2.5	"
	"	"	eyes	0.05 "	0.8	"
	"	"	gills	47.8 "	796.6	"
	"	"	fins	1.46 "	24.3	"
	"	"	scales	2.99 "	49.8	"
	"	"	muscles	0.03 "	0.5	"
	"	"	bones	0.07 "	1.2	"
Nickel (Ni)	8.2±0.2 cpm/g present in soil	Tridacna crocea	kidney	158.0±2.6 cpm/g	...	Beasley and Held 1969[263]
	...	Tridacna crocea	kidney	41.2±0.6 cpm/g	"	"
	80.0±1.0 cpm/g present in soil	"	kidney	163.0±3.0 cpm/g	...	"
Silver (Ag)	7.5 μc injected into air bladder	Crassius auratus	intestine	200–300 cpm/mg	...	Hibiya and Oguri 1961[270]
	"	"	liver	2250 "	...	"
	"	"	pancreas	250–400 cpm/mg	...	"
	"	"	spleen	200–500 "	...	"
	"	"	kidney	~400 "	...	"
	"	"	head kidney	~400 "	...	"
	"	"	gill	~250 "	...	"
	7.5 μc injected into air bladder	Crassius auratus	muscle	100 cpm/mg	...	Hibiya and Oguri 1961[270]
	"	"	backbone	150 "	...	"
	"	"	gonad	200 "	...	"
	"	"	air bladder	500–2500 "	...	"

TABLE 3—Continued

Constituent	Concentration in sea water	Species	Tissue or organ	Concentration in tissue	Concentration factor	Literature Citation
Uranium (U)	3.0×10^{-6} percent	Charaphytae diatomae	whole	2.0×10^{-3} percent U		Kovalsky et al. 1967[290]
	"	fish	whole	6.8×10^{-4} percent U		"
	"	"	boned	$5.4\times10^{-6}-1.2\times10^{-4}$ percent		"
	"	"	kidney	$1.05\times10^{-6}-9.4\times10^{-6}$ percent		"
	"	"	gonads hard roe)	$4.15\times10^{-7}-3.7\times10^{-6}$ percent		"
	"	"	gonads (soft roe)	$2.9\times10^{-7}-1.9\times10^{-6}$ percent		"
	"	"	muscle	$1.37\times10^{-7}-1.32\times10^{-6}$ percent		"
	"	"	blood	$2.2\times10^{-7}-7.0\times10^{-7}$ percent		"
	"	"	brain	$3.22\times10^{-7}-1.0\times10^{-6}$ percent		"
Zinc (Zn)	200.000 cpm l	Meretrix meretrix luzoria	gill	510 cpm g		Saiki and Mori 1955[302]
	"	"	viscera (without liver)	275 "		"
	"	"	mantle	270 "		"
	"	"	liver	245 "		"
	"	"	adductor muscle	165 "		"
	"	"	siphon	165 "		"
	"	"	marginal part of foot	145 "		"
	"	"	central part of foot	140 "		"
	12.000 cpm l	"	ashed soft tissue	15.8 cpm g	1.3	"
	45,000 cpm l (22 days)	Cyprinus carpio	kidney	299 cpm g		"
	"	"	gill	285 "		"
	"	"	scale	65 "		"
	"	"	heart	57 "		"
	"	"	skin	51 "		"
	45.000 cpm l (22 days)	Cyprinus carpio	caudal fin	50 cpm g		Saiki and Mori 1955[302]
	"	"	intestine	27 "		"
	"	"	hepatopancreas	26 "		"
	"	"	vertebrae	5 "		"
	"	"	muscle	3 "		"
	"	"	gall bladder	2 "		"
	45,000 cpm l (3 day)	"	gill	127 "		"
	"	"	skin	0 "		"
	"	"	scale	2 "		"
	"	"	caudal fin	35 "		"
	"	"	vertebrae	0 "		"
	"	"	intestine	27 "		"
	"	"	ga.l bladder	0 "		"
	"	"	hepatopancreas	33 "		"
		"	kidney	89 "		"
	5.000 cpm (45 hrs)	"	gill	119 "		"
	5.000 cpm (45 hrs)	Cyprinus carpio	skin	31 cpm g		Saiki and Mori 1955[302]
	"	"	scale	87 "		"
	"	"	caudal fin	86 "		"
	"	"	vertebrae	29 "		"
	"	"	intestine	121 "		"
	"	"	gall bladder	51 "		"
	"	"	hepatopancreas	251 "		"
	"	"	kidney	1180 "		"
	"	"	heart	173 "		"
	"	"	muscle	9 "		"
	5.000 cpm/l	"	gil	128 "		"
	"	"	skin	40 "		"
	"	"	scale	31 "		"
	"	"	caudal fin	56 "		"
	"	"	vertebrae	36 "		"
	"	"	intestine	50 "		"
	5000 cpm l	Cyprinus carpio	gall bladder	39 cpm g		Saiki and Mori 1955[302]
	"	"	hepatopancreas	65 "		"
	"	"	kidney	690 "		"
	"	"	heart	37 "		"
	"	"	muscle	4 "		"
	20 ppm	Salmo gairdneri	tissue	7.4-12 ppm		Lloyd 1960[291]
	"	"	gills	80-63 ppm		"
	injected dose 9.3 μc	Crassius auratus	intestine	475 cpm/mg	540-4400	Hibiya and Oguri 1961[278]
	"	"	liver	250 "		"
	"	"	pancreas	200 "		"
	"	"	spleen	75 "		"
		"	kidney	130 "		"
	"	"	head kidney	175 "		"

TABLE 3—Continued

Constituent	Concentration in sea water	Species	Tissue or organ	Concentration in tissue	Concentration factor	Literature Citation
Zinc (Zn)	injected dose 9.3 µc	Crassius auratus	gill	110 "		Hibiya and Oguri 1961[270]
	"	"	muscle	30 "		"
	"	"	backbone	75 "		"
	"	"	gonad	40 "		"
		"	air bladder	185 cpm/mg		"
		Clupea harengus	whole	4,400		Ichikawa 1961[284]
		Anguilla anguilla	"	4,200		"
		Mugil cephalus	"	540		"
		Pleuronects sp.	"	2,900		"
		Stichopus tremulus	"	1,400		"
		Palaemon vulgaris	"	1,900		"
		Callinectes hastatus	"	4,400		"
		Octopus vulgaris	"	11,000		"
		Sepia officinalis	"	2,600		"
		Loligo vulgaris	"	5,700		"
		Haliotus tuberculata	"	10,000		"
		Ostrea edulis	"	40,000		"
		Pecten japobaeus	"		17,000	"
		Geldium gracilaria	"		80	"
		Laminario digitata	"		400	"
	(10.3 µc) 0.25 ppm Zn	Ictalurus nebulosus	whole fish	0.045 µg/g		Joyner 1961[287]
	" 0.5 ppm Zn	"	"	0.061 "		"
	" 1.0 ppm Zn	"	"	0.025 "		"
	(3.08 µc) 3.0 ppm Zn	"	"	0.529 "		"
	(61.6 µc) 6.0 ppm Zn	"	"	1.510 "		"
	(10.3 µc) 0.25 ppm Zn	"	"	0.066 "		"
	(10.3 µc) 0.50 ppm Zn	"	"	0.067 "		"
	(10.3 µc) 1.0 ppm Zn	"	"	0.100 "		"
	(30.8 µc) 3.0 ppm Zn	"	"	1.040 "		"
	(61.6 µc) 6.0 ppm Zn	"	"	2.110 "		"
	8.5 µc/l; pH 7.3 26 hrs. in the dark	Porphyra	whole algal disc	230 count/min/g fresh wt.		Gutknecht 1963[274]
	8.5 µc/l; 26 hrs in the light pH 8.6	"	whole algal disc	400 count/min/g fresh wt.		"
	8.5 µc/l; 26 hrs. in the light pH 7.3	"	whole algal disc	300 count/min/g fresh wt.		"
	8.5 µc/l; in 26 hours pH 8.6 in the dark	Porphyra	whole algal disc	330 count/min/g fresh wt.		"
	100 µg/l 15 day exposure	Homarus vulgaris (300 g)	blood	6.7 µg/g wet wt.		Bryan 1964[265]
	"	"	urine	1.7 "		"
	"	"	excretory organs	28.8 "		"
	"	"	abdominal muscle	13.6 "		"
	"	"	hepatopancreas	42.6 "		"
	"	"	stomach fluid	1.1 "		"
	"	"	gills	17.8 "		"
	"	"	shell	11.7 "		"
	100 µg/l 15 day exposure	Homarus vulgaris (300 g)	ovary	30.8 µg/g wet wt.		Bryan 1964[265]
	100 µg/l 42 day exposure	" (390 g)	blood	10.0 "		"
	"	"	urine	40.0 "		"
	"	"	excretory organs	27.8 "		"
	"	"	abdominal muscle	13.3 "		"
	"	"	hepatopancreas	51.9 "		"
	"	"	stomach fluid	0.8 "		"
	"	"	gills	37.5 "		"
	"	"	shells	9.3 "		"
	"	"	vas. deferens	12.0 "		"
	100 µg/l plus 6600 µg Zn over 10 days (injected); 13 days in sea water	" (290 g)	blood	17.5 "		"
		"	urine	4.0 "		"
		"	excretory organs	47.0 "		"
	"	"	abdominal muscle	14.4 "		"
	100 µg/l plus 6600 µg Zn over 10 days (injected); 13 days in sea water	Homarus vulgaris (290 g)	hepatopancreas	158.0 µg/g wet wt.		Bryan 1964[265]
	100 µg/l Zn in sea water plus 6600 µg Zn over 10 days (injected); 3 days after injections	"	stomach fluid	0.7 µg/g wet wt.		"
		"	gills	24.4 "		"
		"	shell	10.1 "		"
	100 µg/l Zn in sea water plus 6600 µg Zn over 10 days (injected); killed 19 days after injections	Homarus vulgaris (460 g)	blood	8.9 "		"
		"	urine	31.8 "		"
		"	excretory organs	24.8 "		"
	"	"	abdominal muscle	12.4 "		"
	"	"	hepatopancreas	117.0 "		"
	"	"	stomach fluid	1.4 "		"
	"	"	gills	29.0 "		"
	"	"	shell	13.8 "		"
	"	"	vas deferens	13.4 "		"

TABLE 3—Continued

Constituent	Concentration in sea water	Species	Tissue or organ	Concentration in tissue	Concentration factor	Literature Citation
Zinc (Zn)	3000 μg Zn injected into stomach; 300 hrs later, 2000 μg Zn; 7 hrs after injection	Homarus vulgaris (300 g)	hepatopancreas	290 μg/g wet wt.		Bryan 1964[265]
		"	blood	27 "		"
	3000 μg injection 150 hrs later	"	excretory organs	117 μg/g wet wt.		"
	"	"	urine	24 μg/g wet wt.		"
	0.004 μc/l 44 days	Paralichthys	whole animal		17	Hoss 1964[283]
	0.4 g radioactive brine-shrimp injested-44 days	"	"		25	"
	2.5 μc/l	Littorina obtusata	whole	6.5×10^{4} cpm/g (3 days)		Mehran and Tremblay 1965[292]
	"	"	"	21 pg/g (of animal)		"
	"	Fucus edentatus	whole	4.2×10^{4} cpm/g (4 days)		"
	7×10^{-9} μc/ml	Carteria sp., Witzschia closterium	whole		(15,900)	Regnier 1965[295]
	"		whole		(13,200)	"
		mullet	"		(230)	"
		mullet	"		(135)	"
	0.002 μc/l	oysters	whole	77±21 μμc/g (1 day)		Duke et al. 1966[272]
	^{65}Zn	mud crabs	"	54±32 " (1 day)		"
	"	clams	"	39±17 " (1 day)		"
	"	snails	"	38±10 " "		"
	"	marsh grass	"	35±11 " "		"
	"	blue crabs	"	32±4 " "		"
	"	mummichogs	"	26±9 " "		"
	"	croakers	"	13±3 " "		"
	"	oysters	"	73±8 " (66 days)		"
	"	mud crabs	"	20±5 " "		"
	"	clams	"	20±6 " "		"
	"	snails	"	20±6 " "		"
	"	marsh grass	"	13±8 " "		"
	"	blue crabs	"	22±1 " "		"
	"	mummichogs	"	18±6 " "		"
		croakers	"	22±2 " "		"
	0.43 c/g ^{65}Zn	Oysters	whole	1,111±502 (1 day)		Duke 1967[271]
	"	Clams	whole	503±166 μμc/g		"
	"	mud clams	whole	853±281 "		"
	0.43 c/g ^{65}Zn	blue crabs	whole	323±172 μμc/g (1 day)		Duke 1967[271]
	"	mummichogs	whole	379±229 "		"
	"	croakers	whole	60±46 "		"
	"	scallops	whole	5,561±578 "		"
	13 μc/l Zn+15 μg/l stable Zn	Ulva pertusa	whole		290 (4 days)	"
	6 μc ^{65}Zn 1,860 μg/l* stable Zn	Vernerupis philippinarum	visceral mass		34 (3 day)	
			shell		4 (3 day)	
	6 μc ^{65}Zn	"	visceral mass		68 (4 days)	"
	60 μg/l stable Zn	"	shell		10	
	13 μc/l ^{65}Zn+15 μg/l stable Zn	Leander sp.	viscera		500	"
			muscle		40	"
			exoskeleton		150	"
	6 μc/l ^{65}Zn+1,860 μg/l stable Zn	Strongylocentrotus pulcherrimus	dig. tract		200 (11 days)	"
			gonad		14 (14 days)	"
			test		10	"
	6 μc/l ^{65}Zn+60 μg/l stable Zn	"	dig. tract		200	"
			gonad		25 (1 day)	"
			test		15	"
	0.1 μC/l ^{65}Zn	Crassostrea virginica	whole		193	Duke et al. 1966[272]
		"	whole		146	"
		"	whole		130	"
		"	whole		139	"
	same as above	Mercenaria mercenaria	whole		18	"
	"	"	whole		22	"
	"	"	"		19	"
	Same as above	Aequipecten irradians	whole		350	"
	"	"	whole		282	"
	"	"	"		243	"
	"	"	"		317	"
	"	Panoplus herbstii	whole		216	"
	"	"	whole		177	"
	"	"	"		166	"
	25 μCi ^{65}Zn/l (1.8 μCi/μg)	Aronyx sp	whole	1.85 μCi/g (120 hr)		Cross et al. 1968[269]
	7.1 μg/l 25 day exposure	Laminaria digitata	whole plant		1800	Bryan 1969[266]
	600 μg/g after 30–31 days uptake	"	"	350 μg/g		"
	2.2 μg/l	"				
	0.028 pCi/ml	Lampsilis radiata	soft tissues	114.2 pCi/g	4080	Harvey 1969[277]
	25 μCi ^{65}Zn/l	Platichthys stellatus	whole	40–90±37 μgZn/g		Renfro and Osterberg 1969[299]
	0.104×10^{-2} pCi	Crassostrea gigas	"	0.99p Ci/g·	5.6×10^{-3}·	Salo and Leet 1969[303]

TABLE 3—Continued

Constituent	Concentration in sea water	Species	Tissue or organ	Concentration in tissue	Concentration factor	Literature Citation
Zinc (Zn)	75 μc/l Zn^{65}	Euphausids	exoskeleton	51.1±10.4 cpm/mg	...	Fowler et al. 1970[279]
	"	"	muscle	27.8±7.0 "	...	"
	"	"	eyes	4.4±1.4 "	...	"
	"	"	haemolymph	16.5±8.3 "	...	"
	75 μc/l Zn^{65}	Prawns-shrimp	exoskeleton	65.8±5.7 cpm/mg	...	Fowler et al. 1970[279]
	"	"	muscle	17.9±4.9 "	...	"
	"	"	hepatopancreas	6.6±3.0 "	...	"
	"	"	eyes	0.9±0.2 "	...	"
	"	"	haemolymph	8.8±3.1 "	...	"
	1.0 μg/l	Crassostrea virginica	soft tissue	159.4±77.6 ppm	1-2×10^6	Wolfe 1970[811]
	"	"	mantle	135 ppm	...	"
	"	"	gills	182 "	...	"
	"	"	labial palps	123 "	...	"
	"	"	muscle	79 ppm	...	"
	"	"	dig. gland	260 "	...	"
	"	"	remainder	175 "	...	"
	"	"	extracellular fluid	6.5 "	...	"
	"	"	pallial fluid	1.2 "	...	"

APPENDIX III–TABLE 4—Maximum Permissible Concentrations of Inorganic Chemicals in Food and Water

Constituent	Maximum Permissible Concentration	Substance Allowed to Contain Given Concentration	Conditions & Comments	Reference
Ammonia (NH_3)	0.5 mg/l	drinking water	recommended limit for domestic water supplies; concentration as $NH_4(+)$	World Health Organization 1961[319] (WHO 1961)
	5 ppm	apples and pears	tolerance for residues ammonium sulfate	Food & Drug Administration 1971[316] (FDA 1971)
Arsenic (As)	3.5 mg/kg	fruits and vegetables	limit for residue on sprayed fruits & vegetables using copper arsenate, calcium arsentae & magnesium arsenate	FDA 1971[316]
	0.1 mg/l	ready-to-drink beverages	limit for content	Food Standards Committee for England & Wales 1959[314]
	1.0 mg/kg	food	regulation on content of food	Food Standards Committee for England & Wales 1959[314]
	0.2 mg/l	drinking water	recommended limit for domestic water supplies; conc. as $NH_4(+)$	WHO 1961[319]
	5 ppm	certain foods	maximum permissible content	Department of National Health & Welfare, Canada 1971 (CANADA 1971)[313]
Barium (Ba)	1.0 mg/l	drinking water	maximum allowable limit	U. S. Department of Health, Education & Welfare, Public Health Service Drinking Water Standards 1962 (PHS 1962)[317]
Boron (B)	30 ppm	cotton seed	residues from post-harvest application	FDA 1971[316]
	8 ppm	citrus fruits	residues from post-harvest application	FDA 1971[316]
Bromine (Br)	75 ppm	vegetables, broccoli, carrots, melons, parsnips, potatoes	tolerance for residues using nematocide ethylene dibromide; concentration as Br	FDA 1971[316]
Bromine (Br)	50 ppm	eggplant, okra, summer squash, sweet corn, sweet potatoes, tomatoes	tolerance for residues using nematocide ethylene dibromide; conc. as Br	FDA 1971[316]
	40 ppm	pineapple	"	"
	30 ppm	cucumber, lettuce, peppers	"	"
	25 ppm	cottonseed, peanuts	"	"
	10 ppm	asparagus, cauliflower	"	"
	5 ppm	lima beans, strawberries	"	"
	50 ppm	cereals	concentration as Br tolerance for residues fumigated after harvest with dibromide	"
	100 ppm	beans, bittermelons, cantalopes, bananas, citrus fruits, cucumber, guavas, litchi fruit, longan fruit, mangoes, papaya, pepper, pineapple, zucchini,	"	"
	25 ppm	cherries and plums	"	"
	75 ppm	malting of barley	Bromate calculated as Br tolerance for residues	"
	325 ppm	parmesan & roquefort cheese	residues for Bromides calculated as Br	"
	400 ppm	dried eggs, processed herbs and spices	"	"
Bromine (Br)	25 ppm	raspberries, summer squash	tolerance for residues of inorganic bromides; concentration as Br	FDA 1971[316]
	20 ppm	citrus fruit	"	"
	15 ppm	cherries and plums	"	"
	10 ppm	walnuts and strawberries	"	"
	5 ppm	apricots, nectarines, peaches	"	"
	60 ppm	eggplant	"	"
	40 ppm	muskmelon, tomato	"	"
	25 ppm	broccoli, cauliflower, peppers, pineapples, strawberries	"	"
	400 ppm	dog food	"	"
	125 ppm	cereals	"	"
	90 ppm	dehydrated citrus fruit for cattle	soil treatment with nematocide 1,2-dibromo 3,chloropropane tolerance for residues calculated as Br	"
	130 ppm	endive and lettuce	"	"
	125 ppm	bananas	tolerance for residues calculated as Br	"
	75 ppm	almond hulls, carrots, celery, snap beans, turnip	"	"
Bromine (Br)	50 ppm	almonds, brussel sprouts, broccoli, cabbage, cauliflower, eggplant, melon, peanuts, peppers, pineapples, tomatoes	tolerance for residues calculated as Br	FDA 1971[316]
	25 ppm	berries, cottonseed, cucumbers, grapes	"	"
Cadmium (Cd)	0.1 mg/l	drinking water	maximum permissible concentration of Cd in domestic supplies	Kirkor 1951[315]
	0.01 mg/l	drinking water	mandatory limit of Cd in domestic supplies	PHS 1962[317]
	0.05 mg/l	drinking water	tolerance limit of Cd in domestic supplies	WHO 1961[319]
Calcium (Ca)	75 mg/l	drinking water	permissible limit	World Health Organization International Standards for Drinking Water 1958 (WHO 1958)[318]
Chromium (Cr)	0.05 mg/l	drinking water	mandatory limit for Cr^{+6} in domestic supplies	PHS 1962[317]
	0.05 mg/l	drinking water	mandatory limit	WHO 1961[319]
	0.05 mg/in²	for covering surface of food containers	limit not to be exceeded	FDA 1971[316]
	7 µg/in²	closure area of packing containers	concentration calculated as Cr using chromic chloride complexes	FDA 1971[316]
Copper (Cu)	1.0 mg/l	drinking water	recommended limit	PHS 1962[317]
	1.0 mg/l	drinking water	permissible limit for domestic water supplies	WHO 1958[318]
	0.05 mg/l	drinking water	permissible limit for domestic water supplies	WHO 1961[319]
	2 mg/l	ready-to-drink beverages	established limits	British Ministry of Agriculture, Fisheries and Food 1956[313]
	7 mg/l	cider and concentrated soft drinks	established limits	"
	20 mg/kg	most foods	established limits	"
	60 mg/kg	yeast and yeast products	"	"
	300 mg/kg	solid pectin	"	"

TABLE 4—Continued

Constituent	Maximum Permissible Concentration	Substance Allowed to Contain Given Concentration	Conditions & Comments	Reference
Copper (Cu)	3 ppm	pears	tolerance for residues complexed copped for copper carbonate; post-harvest use	FDA 1971[316]
	100 ppm	certain foods	maximum quantities	CANADA 1971[313]
Cyanide (CN)	0.01 mg/l	drinking water	maximum allowable limit	WHO 1958,[318] 1961[319]
	0.01 mg/l	drinking water	recommended limit	PHS 1962[317]
	0.2 mg/l	drinking water	mandatory limit	"
	25 ppm	cereals and grains	post-harvest application of CaCN	FDA 1971[316]
	250 ppm	spices	post-harvest fumigation with HCN; tolerances for residues	FDA 1971[316]
	100 ppm	cereals	"	"
	25 ppm	nuts, i.e. almonds, etc.	"	"
	125 ppm	cereal flours	limits not to be exceeded	"
	90 ppm	cereals cooked before eating	residues of HCN shall not exceed these limits	"
	50 ppm	uncooked pork	"	"
	20 ppm	cocoa	"	"
	0.15 percent	bakery products	"	"
	0.1 percent	egg white solids	"	"
	0.095 percent	frozen meat	"	"
	0.15 percent	yeast	"	"
Fluorine (F)	1.2 mg/l	drinking water	recommended control limits optimum; 50-53.7 F	PHS 1962[317]
	0.7 mg/l	drinking water	at 79.3-90.5 F	"
	1.5 mg/l	drinking water	recommended limit	WHO 1961[319]
	7 ppm	apple, apricot, bean, beet, blackberries, blueberries, boysenberries, broccoli, brussel sprout, etc. most fruits & vegetables	tolerance of combined fluorine for insecticidal fluorine compounds, cryolite and synthetic cryolite	FDA 1971[316]
Fluorine (F)	25 ppm	certain foods	maximum	CANADA 1971[313]
Iron (Fe)	0.3 mg/l	drinking water	recommended limit	PHS 1962[317]
	0.3 mg/l	drinking water	permissible limit	WHO 1958[318]
	1.0 mg/l	drinking water	excessive limit	"
	0.1 mg/l	drinking water	recommended limit	WHO 1961[319]
Lead (Pb)	10 ppm	certain foods	maximum permissible levels mandatory limit for domestic water supplies	CANADA 1971[313]
	0.05 mg/l	drinking water	"	PHS 1962[317]
	0.1 mg/l	drinking water	"	WHO 1958,[318] 1961[319]
	7 ppm	most fruit, i.e. apples, grapes, mangoes, peaches, cherries, etc; tomatoes, young berries, raspberries, peppers, etc.	tolerance of combined lead using lead arsenate	FDA 1971[316]
Magnesium (Mg)	125 mg/l	drinking water	recommended limit for domestic water supply	WHO 1961[319]
Manganese (Mn)	0.05 mg/l	drinking water	recommended limit for domestic water supply	PHS 1962[317]
	0.10 mg/l	drinking water	permissible limit for domestic water supply	WHO 1958[318]
	0.50 mg/l	drinking water	excessive limit for domestic water supply	"
Manganese (Mn)	0.01 mg/l	drinking water	recommend limit for domestic water supply	WHO 1961[319]
Mercury (Hg)	0.005 mg/l	drinking water	maximum permissible concentration	Kirkor 1951[315]
	0.5 ppm	certain foods	interim guidelines	CANADA 1971[313]
Nickel (Ni)	1.0 mg/l	drinking water	maximum permissible concentration	Kirkor 1951[315]
Nitrates	50 mg/l	drinking water	recommended limit for domestic water supply	WHO 1961[319]
Selenium (Se)	0.01 mg/l	drinking water	mandatory limit for domestic water supply	PHS 1962[317]
	0.05 mg/l	drinking water	"	WHO 1958,[318] 1961[319]
Silver (Ag)	0.05 mg/l	drinking water	"	PHS 1962[317]
Zinc (Zn)	5 mg/l	drinking water	recommended limit for domestic water supply	"
	5 mg/l	drinking water	"	WHO 1958,[318] 1961[319]
	65 ppm	peanut, vine hay & sugar beets	using Zn ion calculated as Zn	FDA 1971[316]
	25 ppm	straws of barley oats & rye, wheat	"	"
	15 ppm	bananas, fodder of field corn, sweet corn and popcorn	pre & post-harvest use, Zn ion calculated as Zn	"
Zinc (Zn)	10 ppm	apples, celery, crabapples, fennel, pears, quinces, papayas	pre and post-harvest use, Zn ion calculated as Zn	FDA 1971[316]
	7 ppm	cranberries, cucumbers, grapes, summer squash, tomatoes, melons	using Zn ion calculated as Zn	"
	5 ppm	grains of barley, oats, rye and wheat	"	"
	2 ppm	carrots, sugar beets	"	"
	0.5 ppm	corn, grain, cotton seed, kidney, liver, onions, peanuts	"	"
	0.1 ppm	asparagus	"	"
	30 ppm	peaches	tolerance for residues of fungicide basic zinc sulfate	"

APPENDIX III-TABLE 5—Total Annual Production of Inorganic Chemicals in the U.S.A.

(U.S. Department of Commerce, Bureau of the Census, 1971)[22]

Constituent	Form of Element	Total Annual Production (short tons)	Product Code	Year
Aluminum (Al)	Al_2O_3—100 percent	6,639,891	2819511	1969
	$AlCl_3$—liquid & crystal	23,838	2819611 & 2819615	1969
	$AlCl_3$—(100 percent) anhydrous	38,511	2819617	1969
	$Al_2O_3\cdot 3\,H_2O$—100 percent	325,767	2819625	1969
	AlF_3—(tech)	143,131	2819627	1969
	$Al_2(SO_4)_3$—(comm) 17 percent Al_2O_3	1,243,803	2819651	1969
Ammonia (NH_3)	synthetic—anhydrous	12,917,842	2819131	1969
	byproduct liquor	14,000	2819131	1969
	NH_4Cl—gray & white	26,615	2819141 & 2819143	1965*
	NH_4NO_3—100 percent	6,891,234	2819151	1969
	$(NH_4)_2SO_4$—100 percent	1,915,721	2819157	1969
Barium (Ba)	$BaCO_3$—100 percent	79,002	2819904	1969
Bismuth (Bi)	subcarbonate 100 percent $(Bi_2O_2CO_3)\cdot H_2O$	57	28199—	1969
Boron	boric acid—100 percent	138,969	2819411	1969
	$Na_2B_4O_7\cdot 10\,H_2O$	624,257	2819724	1969
Calcium (Ca)	carbide—(Comm)	856,039	2819912	1969
	$CaHPO_4$—animal feed grades 100 percent	496,027	2819919	1969
	$CaHPO_4$—other grades	416,096	2819920	1969
	$CaCO_3$—100 percent	206,078	2819913	1969
Chlorine (Cl)	100 percent Cl_2 gas	9,413,885	2812111	1969
	100 percent Cl_2 liquid	4,399,712	2812115	1969
	calcium hypochlorite (75 percent Cl)	42,941	2819211	1969
	HCl—100 percent	1,910,757	2819441, 2819445 & 2819447	1969
	$NaClO_3$—100 percent	187,221	2819727	1969
Chromium (Cr)	chromic acid—100 percent	24,858	2819431	1969
	sodium bichromate and chromate	132,593	2819929 & 2819931	1969
Copper (Cu)	CuO—100 percent	1,910	2819934	1968*
	Cu_2O—100 percent	1,742	2819935	1969
	$CuSO_4\cdot 5\,H_2O$—100 percent	47,163	2819936	1969
Cyanide (CN)	HCN—100 percent	205,208	2819451	1969
Fluorine (F)	HF—100 percent anhydrous	221,536	2819461	1969
	NaF—100 percent	6,885	2819728	1969
	Na_2SiF_6—100 percent	48,975	2819751	1969
	HF—100 percent	17,206	2819465	1969
Hydrogen (H^+)	H_2SO_4—100 percent	29,536,914	28193—	1969
Iron (Fe)	$FeCl_3$—100 percent	66,674	2819942	1969
	$FeSO_4$—100 percent	192,020	2819943	1969
Manganese (Mn)	$MnSO_4\cdot 4\,H_2O$	40,806	2819950	1969
Mercury (Hg)	mercury—redistilled	475,688 (lbs)	2819953	1969
Nickel (Ni)	$NiSO_4\cdot 6\,H_2O$—100 percent	20,388	2819956	1969
Phosphorus (P)	elemental—white & red (tech)	628,957	2819958 & 2819959	1969
	$POCl_3$—100 percent	31,404	2819960	1969
	P_2S_5—100 percent	55,759	2819961	1969
	P_2O_5—100 percent	3,566	2819962	1966*
	PCl_3—100 percent	57,312	2819963	1969
Silver (Ag)	AgCN—100 percent	1,795 (thousand av oz.)	2819971	1969
	$AgNO_3$—	113,809	2819972	1969**
Sulphide	NaSH—100 percent	27,364	2819729	1969
	Na_2S—60-62 percent concentrated	22,222	2819782	1967***
	Na_2S—60-62 percent concentrated crystal & liquid (total)	122,022	2819781, 2819782, & 2819783	1969
Zinc (Zn)	$ZnCl_2$—100 percent	27,986	2819984	1966*
	$ZnSO_4\cdot 7\,H_2O$—100 percent	57,774	2819987	1969

* 1965 through 1968 figures withheld to avoid disclosing the figures for individual companies.

** Includes unspecified amounts produced and shipped on contract basis.

*** combined with 81 and 83 for 1968.

APPENDIX III-TABLE 6—Toxicity

Substance Tested	Formulation	Organism Tested	Common Name	Life Stage or Size (mm)	Conc. (ppb act. ingred.) in water	Methods of Assessment
Insecticides Organochloride						
Aldrin	Technical	Palaemon macrodactylus*	Korean shrimp		.74 (.51–1.08)	TL-50
Aldrin	Technical	Palaemon macrodactylus	Korean shrimp		3 (1.1–8.5)	TL-50
Aldrin	100 percent	Crangon septemspinosa	Sand shrimp	26	8	LC-50
Aldrin	100 percent	Palaemonetes vulgaris	Grass shrimp	31	9	LC-50
Aldrin	100 percent	Pagurus longicarpus	Hermit crab	3.5	33	LC-50
Aldrin		Mercenaria mercenaria	Hard clam	Larvae	500	37 percent survival
				Larvae	1000	00 percent
				Eggs	>10000	TLM
				Larvae	410	TLM
Aldrin	100 percent	Fundulus heteroclitus	Mummichog	42	4	LC-50
Aldrin	100 percent	Fundulus heterocktus	Mummichog	55	8	LC-50
Aldrin	100 percent	Fundulus majalis	Striped killifish	49	17	LC-50
Aldrin	100 percent	Menidia menidia	Atlantic silverside	57	13	LC-50
Aldrin	100 percent	Mugil cephalus	Striped mullet	85	100	LC-50
Aldrin	100 percent	Thalassoma bifasciatum	Bluehead	80	12	LC-50
Aldrin	100 percent	Sphaeroides maculatus	Northern puffer	168	36	LC-50
Aldrin	100 percent	Anguilla rostrata	American eel	56	5	LC-50
Aldrin	Technical	Gasterosteus aculeatus	Threespine stickleback	22–44	27.4	TLM
Aldrin	Technical	Cymatogaster aggregata	Shiner perch		7.4	TL-50
Aldrin	Technical	Cymatogaster aggregata	Shiner perch		2.26 (1.08–4.74)	TL-50
Aldrin	Technical	Micrometrus minimus	Dwarf perch		18	TL-50
Aldrin	Technical	Micrometrus minimus	Dwarf perch		2.03 (1–4.2)	TL-50
Tri-6-Dust	81 percent	Penaeus setiferus	White shrimp	41.6±5.9	35[a]	TLM
	Benzene Hexachoride	Penaeus aztecus	Brown shrimp	11.9±.45	400[a]	TLM
Chlordane	100 percent	Palaemon macrodactylus	Korean shrimp		18 (10–38)	TL-50
Chlordane	100 percent	Palaemon macrodactylus	Korean shrimp		11 (7–18)	TL-50
Chlordane	100 percent	Gasterosteus aculeatus	Threespine stickleback	22–44	160.	TLM
DDT		Dunaliella euchlora			1000	42 percent reduction in O_2 evolution
DDT		Dunaliella euchlora			100	32 percent reduction in O_2 evolution
DDT		Dunaliella euchlora			10	30 percent reduction in O_2 evolution
DDT		Phaeodactylum tricornutum			1000	35 percent reduction in O_2 evolution
DDT		Skeletonema costatum			1000	39 percent reduction in O_2 evolution
DDT		Skeletonema costatum			100	32 percent reduction in O_2 evolution
DDT		Skeletonema costatum			10	36 percent reduction in O_2 evolution
DDT		Cyclotella nana			1000	33 percent reduction in O_2 evolution
DDT		Cyclotella nana			100	33 percent reduction in O_2 evolution
						Effect of toxicant on growth of phytoplankton
DDT	Wettable powder	Protococcus sp			600	0.50 value a
DDT	Wettable powder	Chlorella sp			600	1.00 ratio of O.D.
DDT	Wettable powder	Dunaliella euchlora			600	0.74 of Expt./O.D.
DDT	Wettable powder	Phaeodactylum tricornutum			600	0.91 control
DDT	Wettable powder	Monochrysis lutheri			40	0.57
DDT[b]		Crassostrea virginica	American oyster	27 mean height	1.0	Weight difference between control and experimental oysters
Toxaphene					1.0	
Parathion					1.0	
DDT	Technical Grade	Penaeus duorarum	Pink shrimp	13.3 mm (Aug.)	0.12	TL-50
DDT	77 percent	Palaemon macrodactylus	Korean shrimp		0.86 (0.47–1.59)	TL-50
DDT	77 percent	Palaemon macrodactylus	Korean shrimp		0.17 (0.09–0.32)	TL-50
DDT	P, P¹ isomer	Crangon septemspinosa	Sand shrimp	26	0.6	LC-50
DDT	P, P¹ isomer	Palaemonetes vulgaris	Grass shrimp	31	2.	LC-50
DDT		Callinectes sapidus	Blue crab	Adult	19. (9.–36.)	TLM
DDT		Callinectes sapidus	Blue crab	Adult	35. (21–57)	TLM
DDT	P, P¹ isomer	Pagurus longicarpus	Hermit crab	3.5	6	LC-50
DDT	P, P¹ isomer	Fundulus heteroclitus	Mummichog	42	3	LC-50
DDT	P, P¹ isomer	Fundulus heteroclitus	Mummichog	55	5	LC-50
DDT	P, P¹ isomer	Fundulus majalis	Striped killifish	40	1	LC-50
DDT	P, P¹ isomer	Menidia menidia	Atlantic silverside	59	.4	LC-50
DDT	P, P¹ isomer	Mugil cephalus	Striped mullet	46	.9	LC-50
DDT	P, P′ isomer	Anguilla rostrata	American eel	56 mm	4	LC-50
DDT	P, P′ isomer	Thalassoma bifasciatum	Bluehead	80 mm	7	LC-50
DDT	P, P′ isomer	Sphaeroides maculatus	Northern puffer	140 mm	89.	LC-50

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* N.B. Italic type fonts were not available in a suitable point size. Ed.

[a] Concentration of Tri-6-dust.

Data for Organic Compounds

Test Procedure	Temp C	Salinity ‰	Other Environmental Criteria	Statistical Evaluation	Residue levels mg/kg	Other Parameters	Reference
96 hr static lab bioassay	13-18	12-30	Turb. 1-12 JTU	95 percent confidence intervals			Earnest (unpublished)[353]
96 hr intermittent flow lab bioassay	13-18	12-30	Turb. 1-12 JTU				″
96 hr static bioassay	20±.5	24	pH 8.0 D.O. 7.1-7.7 mg/l				Eisler 1969[327]
96 hr static bioassay	20±.5	24	pH 8.0 D.O. 7.1-7.7 mg/l				Eisler 1969[327]
96 hr static bioassay	20±.5	24	pH 8.0 D.O. 7.1-7.7 mg/l				Eisler 1969[327]
10 day two-cell stage fertilized	24±1						Davis and Hidu 1969[326]
10 day eggs introduced into test media	24±1						Davis and Hidu 1969[326]
48 hr 50 percent of eggs develop normally	24±1						Davis and Hidu 1969[326]
12 day 50 percent of larvae survive	24±1						Davis and Hidu 1969[326]
96 hr static bioassay	20	24	pH 8.0 D.O. 7.1-7.7 mg/l				Eisler 1970a[328]
96 hr static bioassay	20	24	pH 8.0 D.O. 7.1-7.7 mg/l				Eisler 1970b[329]
96 hr static bioassay	20	24	pH 8.0 D.O. 7.1-7.7 mg/l				Eisler 1970b[329]
96 hr static bioassay	20	24	pH 8.0 D.O. 7.1-7.7 mg/l				Eisler 1970b[329]
96 hr static bioassay	20	24	pH 8.0 D.O. 7.1-7.7 mg/l				Eisler 1970b[329]
96 hr static bioassay	20	24	pH 8.0 D.O. 7.1-7.7 mg/l				Eisler 1970b[329]
96 hr static bioassay	20	24	pH 8.0 D.O. 7.1-7.7 mg/l				Eisler 1970b[329]
96 hr static bioassay	20	24	pH 8.0 D.O. 7.1-7.7 mg/l				Eisler 1970b[329]
96 hr static bioassay	20±.5	25	pH 6.8-7.4 Total alkalinity= 45-57 ppm				Katz 1961[333]
96 hr static bioassay	13±1	28	5.0 JTU Turbidity	95 percent confidence intervals			Earnest and Benville (unpublished)[364]
96 hr intermittent flow bioassay	14-18	25(25-26)	Turbidity 7 (5-10) JTU		1.0(0.23-0.42)		″
96 hr static lab bioassay	13±1	18	Turbidity 18 JTU	″	0.38(0.22-0.54)		″
96 hr intermittent flow bioassay	14-18	28	Turbidity 7 JTU				″
24 hr static lab bioassay	17.4-22.3	31.4	pH = 8.15-8.2	None			Chin and Allen 1957[323]
24 hr static lab bioassay	17.4-22.3	31.4	pH = 8.15-8.2	None			Chin and Allen 1957[323]
96 hr static lab bioassay	13-18	12-30	Turb. 1-12 JTU	95 percent confidence intervals			Earnest (unpublished)[353]
96 hr intermittent flow lab bioassay	13-18	12-30	Turb. 1-12 JTU	95 percent confidence intervals			Earnest (unpublished)[353]
96 hr static lab bioassay	20±.5	25	pH 6.8-7.4 Total Alkalinity as CaCo$_3$ 45-57 ppm	None			Katz 1961[333]
				t-values			
O_2 production measured by Winkler Light-and-Dark Bottle Technique. Length of test 4 hr.			250 ft.-c for 4 hrs	Results analyzed using 1-tailed T-test significance at 0.05 level 8.4			Darby and Ruber 1971[325]
				5.3			″
				5.3			″
				2.9			″
				5.5			″
				2.5			″
				5.2			″
				2.9			″
				2.3			″
Organisms grown in test media containing pesticides for 10 days O.D. measured at 530 mμ	20.5±1	22-28	500 ft.-c continuous	None			Ukeles 1962[347]
	20.5±1	22-28		″			″
	20.5±1	22-28		″			″
	20.5±1	22-28		″			″
	20.5±1	22-28		″			″
36 wk chronic lab bioassay	9-25	27-29		Mean in water weights were statistically different at 0.05 after 22 wks.	DDE = 13.0[c] DDE = .20 DDT = 29.0 Toxaphene = 9.0 Parathion = .007	Tissue changes associated with gill, kidneys, digestive tubules, visceral ganglion and tissues beneath gills. Mycelial fungus also present.	Lowe et al. 1971b[341]
28 day flowing lab bioassay	21-29	24-33		95 percent confidence intervals	0.19 (muscle)		Nimmo et al. (unpublished)[366]
96 hr static lab bioassay	13-18	12-30	Turb. 1-12 JTU	95 percent confidence intervals			Earnest (unpublished)[353]
96 hr flowing lab bioassay	13-18	12-30	Turb. 1-12 JTU	95 percent confidence intervals			Earnest (unpublished)[353]
96 hr static lab bioassay	20	24	pH = 8.0				Eisler 1969[327]
96 hr static lab bioassay	20	24	pH = 8.0	None SL			Eisler 1969[327]
96 hr static lab bioassay	10	8.6		95 percent confidence 1.5			Mahood et al. 1970[342]
96 hr static lab bioassay	21	8.6		Interval slope func. 1.9			Mahood et al. 1970[342]
96 hr static lab bioassay	20	2.4	pH = 8.0	None			Eisler 1969[327]
96 hr static lab bioassay	20	24	pH = 8.0	None			Eisler 1970a[328]
96 hr static lab bioassay	20	24	pH = 8.0	None			Eisler 1970b[329]
96 hr static lab bioassay	20	24	pH = 8.0	None			Eisler 1970b[329]
96 hr static lab bioassay	20	24	pH = 8.0	None			Eisler 1970b[329]
96 hr static lab bioassay	20	24	pH = 8.0	None			Eisler 1970b[329]
96 hr static lab bioassay	20	24	pH = 8.0 DO 7.1-7.8	None			Eisler 1970b[329]
96 hr static lab bioassay	20	24	pH = 8.0 DO 7.1-7.8	None			Eisler 1970b[329]
96 hr static lab bioassay	20	24	pH = 8.0 DO 7.1-7.8	None			Eisler 1970b[329]

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[b] Mixture of 1.0 ppb of DDT, Toxaphene, Parathion.

[c] Residue after 36 week exposure.

TABLE 6—

Substance Tested	Formulation	Organism Tested	Common Name	Life Stage or Size (mm)	Conc. (ppb act. ingred.) in water	Methods of Assessment
DDT	P, P′ isomer	Gasterosteus aculeatus	Threespine stickleback	22–44 mm	11.5	TLM
DDT	Technical grade	Cymatogaster aggregata	Shiner perch	48–104	7.6	TL-50
DDT	Technical grade	Micrometrus minimus	Dwarf perch	48–104	4.6	TL-50
DDT	P, P′ isomer	Cymatogaster aggregata	Shiner perch		.45 (0.21–0.94)	TL-50
DDT	P, P′ isomer	Micrometrus minimus	Dwarf perch		0.26 (0.13–0.52)	TL-50
Dieldrin	85 percent	Palaemon macrodactylus	Korean shrimp		16.9 (10.9–33.4)	
Dieldrin	85 percent	Palaemon macrodactylus	Korean shrimp		6.9 (3.7–13.1)	
Dieldrin	100 percent	Crangon septemspinosa	Sand shrimp	26 mm	7	LC-50
Dieldrin	100 percent	Palaemonetes vulgaris	Grass shrimp	31 mm	50	LC-50
Dieldrin	100 percent	Pagurus longicarpus	Hermit crab	3.5 mm	18	LC-50
Dieldrin		Crassostrea virginica	American oyster	Egg	640.	TLM
Dieldrin		Nassa obsoleta	Mud snail	Adult	1,000	No. egg cases deposited significant less than control. Control=1473 Expt.=18
Dieldrin		Fundulus heteroclitus	Mummichog	37 mm	5	LC-50
Dieldrin	100 percent	Fundulus heteroclitus	Mummichog	51 mm	5	LC-50
Dieldrin	100 percent	Fundulus majalis	Striped killifish	40 mm	4	LC-50
Dieldrin	100 percent	Menidia menidia	Atlantic silverside	57 mm	5	LC-50
Dieldrin	100 percent	Mugil cephalus	Striped mullet	85 mm	23	LC-50
Dieldrin	100 percent	Anguilla rostrata	American eel	57 mm	.9	LC-50
Dieldrin	100 percent	Thalassoma bifasciatum	Bluehead	80 mm	6	LC-50
Dieldrin	100 percent	Sphaeroides maculatus	Northern puffer	168 mm	34.	LC-50
Dieldrin	Technical	Gasterosteus aculeatus	Threespine stickleback	22–44	13.1	TLM
Dieldrin	Technical	Cymatogaster aggregata	Shiner perch	48–104	3.7	TL-50
Dieldrin	Technical	Micrometrus minimus	Dwarf perch	48–104	5.	TL-50
Dieldrin	0.012 percent W/V	Poecilia latipinna	Sailfin mollie	?	7.5	Reduced reproduction control—young born 65 Exp.—young born 37
Dieldrin	0.012 percent W/V	Poecilia latipinna	Sailfin mollie		6. 12.	SGOT activity[a] increase
Dieldrin	Technical	Cymatogaster aggregata	Shiner perch		1.5 (0.73–3.20)	TL-50
Dieldrin	Technical	Micrometrus minimus	Dwarf perch		2.44 (1.16–5.11)	TL-50
Endrin	99 percent	Palaemon macrodactylus	Korean shrimp		4.7 (2.3–9.4)	TL-50
Endrin	99 percent	Palaemon macrodactylus	Korean shrimp		.12 (0.05–0.25)	TL-50
Endrin	100 percent	Crangon septemspinosa	Sand shrimp	26 mm	1.7	LC-50
Endrin	100 percent	Palaemonetes vulgaris	Grass shrimp	31 mm	1.8	LC-50
Endrin	100 percent	Pagurus longicarpus	Hermit crab	3.5	12	LC-50
Endrin	100 percent	Nassa obsoleta	Mud snail	Adult	1,000	No. egg cases deposited significantly less than Control. Control=1473 Expt.=2
Endrin		Crassostrea virginica	American oyster	Egg	790	TLM
Endrin		Fundulus heteroclitus	Mummichog	42 mm	0.6	LC-50
Endrin	100 percent	Fundulus heteroclitus	Mummichog	51 mm	.6	LC-50
Endrin	100 percent	Fundulus majalis	Striped killifish	40 (mm)	0.3	LC-50
Endrin	Technical 98 percent	Fundulus similis	Longnose killifish		0.23	LC-50
Endrin	Technical 98 percent	Brevoortia patronus	Menhaden		0.8	LC-50
Endrin	Technical 98 percent	Mugil cephalus	Striped mullet		2.6	LC-50
Endrin	100 percent	Mugil cephalus	Striped mullet	83 (mm)	0.3	LC-50
Endrin	100 percent	Menidia menidia	Atlantic silverside	54 (mm)	0.05	LC-50
Endrin	100 percent	Thalassoma bifasciatum	Bluehead	90 (mm)	0.1	LC-50
Endrin	100 percent	Anguilla rostrata	American eel	57 (mm)	0.6	LC-50
Endrin	100 percent	Sphaeroides maculatus	Northern puffer	131 (mm)	3.1	LC-50
Endrin	Technical 90 percent	Gasterosteus aculeatus	Threespine stickleback	22–44	0.5	TLM
Endrin	Powder 75 percent	Gasterosteus aculeatus	Threespine stickleback	25–37	1.5	TLM
Endrin	Technical 98 percent	Cyprinodon vareigatus	Sheepshead minnow		0.32	LC-50
Endrin	Technical 98 percent	Leiostomus xanthurus	Spot		0.45	LC-50
Endrin	Technical	Cymatogaster aggregata	Shiner perch	48–104	0.8	TLM
Endrin	Technical	Micrometrus minimus	Dwarf perch	48–104	0.6	TLM
Endrin	Technical	Cymatogaster aggregata	Shiner perch	48–104	0.12 (0.06–0.25)	TLM
Endrin	Technical	Micrometrus minimus	Dwarf Perch	48–104	0.13 (0.06–0.27)	TLM
Heptachlor	99 percent	Palaemon macrodactylus	Korean shrimp		14.5 (8.2–25.9)	TL-50
Heptachlor	100 percent	Crangon septemspinosa	Sand shrimp	26	8	LC-50
Heptachlor	100 percent	Palaemonetes vulgaris	Grass shrimp	31	440	LC-50
Heptachlor	100 percent	Pagurus longicarpus	Hermit crab	3.5	55	LC-50
Heptachlor	100 percent	Fundulus heteroclitus	Mummichog	42	50	LC-50
Heptachlor	100 percent	Fundulus heteroclitus	Mummichog	35	50.0	LC-50
Heptachlor	100 percent	Fundulus majalis	Striped killifish	40	37	LC-50
Heptachlor	100 percent	Menidia menidia	Atlantic silverside	54	3	LC-50
Heptachlor	100 percent	Mugil cephalus	Striped mullet	100	194	LC-50
Heptachlor	100 percent	Anguilla rostrata	American eel	56	10	LC-50
Heptachlor	100 percent	Thalassoma bifasciatum	Bluehead	80	.8	LC-50

Continued

Test Procedure	Temp C	Salinity °/₀₀	Other Environmental Criteria	Statistical Evaluation	Residue levels mg/kg	Other Parameters	Reference
96 hr static lab bioassay	20±.5	25	Tot. Alk. as CaCo$_3$ 24-57 ppm	None			Katz 1961[333]
96 hr static lab bioassay	13±1	26	pH 6.8-7.4				Earnest and Benvile (unpublished)[334]
96 hr static lab bioassay	13±1	28		95 percent confidence intervals	0.55(.44-.65) ppm		Earnest and Benville (unpublished)[334]
96 hr inter. flow lab bioassay	14-18	18	Turb. 12 JTU	95 percent confidence intervals	1.0(0.48-2.0) ppm		Earnest and Benville (unpublished)[334]
96 hr inter. flow lab bioassay	14-18	27	Turb 4 JTU				″
96 hr static lab bioassay	13-18	12-30	Turb 1-12 JTU	95 percent confidence intervals			Earnest (unpublished)[332]
96 hr inter. flow lab bioassay	13-18	12-30	Turb 1-12 JTU	95 percent confidence intervals			Earnest (unpublished)[332]
96 hr static lab bioassay	20	24	pH=8.0	None			Eisler 1969[327]
96 hr static lab bioassay	20	24	pH=8.0	None			Eisler 1969[327]
96 hr static lab bioassay	20	24	pH=8.0	None			Eisler 1969[327]
48 hr static lab bioassay	24±1	?					Davis and Hidu 1969[321]
96 hr exposure to 1.0 ppm then 133 day post exposure in clean water	20±.5	24	pH=8.0	No. of egg cases deposited significantly different at 0.001 level			Eisler 1970c[330]
96 hr static lab bioassay	20±.5	24	pH=8.0	None			Eisler 1970a[328]
96 hr static lab bioassay	20±.5	24	pH=8.0	None			Eisler 1970b[329]
96 hr static lab bioassay	23±.5	24	pH=8.0	None			Eisler 1970b[329]
96 hr static lab bioassay	23±.5	24	pH=8.0	None			Eisler 1970b[329]
96 hr static lab bioassay	23±.5	24	pH=8.0	None			Eisler 1970b[329]
96 hr static lab bioassay	23±.5	24	pH=8.0	None			Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24	pH=8.0	None			Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24	pH=8.0	None			Eisler 1970b[329]
96 hr static lab bioassay	20±.5	25	pH 6.8-7.4 Tot. Alk CaCo$_3$ 45-57 ppm	None			Katz 1961[333]
96 hr static lab bioassay	13±1	15		None			Earnest and Benville (unpublished)[334]
96 hr static lab bioassay	13±1	29		None			Earnest and Benville ″
34 wk flowing water	17-30	25-30		None	Blood 11.98 Brain 13.3 Gill 37.6 ppm		Lane and Livingston 1970[335]
48 hr flowing water test	27±1	7		Activity significantly greater at 0.05 level			Lane and Scura 1970[336]
96 hr inter. flow lab bioassay	14-18	28	Turb 6 JTU	95 percent confidence interval	2.33(0.00168-0.00307) ppm		Earnest and Benville (unpublished)[334]
96 hr inter. flow lab bioassay	14-18	12	Turb 24 JTU	95 percent confidence interval	1.26(0.00086-0.0017)		Earnest and Benville ″
96 hr static lab bioassay	13-18	12-30	Turb 1-12 JTU	95 percent confidence interval			Earnest (unpublished)[332]
96 hr inter. flow lab bioassay	13-18	12-30	Turb 1-12 JTU	95 percent confidence interval			Earnest ″
96 hr static lab bioassay	20	24	pH=8.0	None			Eisler 1969[327]
96 hr static lab bioassay	20	24	pH=8.0	None			Eisler 1969[327]
96 hr static lab bioassay	20	24	pH=8.0	None			Eisler 1969[327]
96 hr static exposure of adults to 1.0 ppm. 133 day post exposure in clean water	20±.5	24		No. of egg cases deposited significantly different at 0.001 level			Eisler 1970c[330]
48 hr static lab bioassay	24±1			None			Davis and Hidu 1969[321]
96 hr static lab bioassay	20±.5	24	pH=8.0	None			Eisler 1970a[328]
96 hr static lab bioassay	20±.5	24	pH=8.0	None			Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24		None			Eisler 1970b[329]
24 hr flowing lab bioassay	25	19		None			Lowe 1965[338]
24 hr flowing lab bioassay	27	29		None			Lowe 1965[338]
24 hr flowing lab bioassay	29	21		None			Lowe 1965[338]
96 hr static lab bioassay	20±.5	24		None			Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24		None			Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24		None			Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24		None			Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24		None			Eisler 1970b[329]
96 hr static lab bioassay	20±.5	25	pH=6.8-7.4 Tot. Alk. as (CaCO$_3$) 45-57	None			Katz 1961[333]
96 hr static lab bioassay	20	25		None			Katz and Chadwick 1961[334]
24 hr flowing lab bioassay	28	29		None			Lowe 1965[338]
24 hr flowing lab bioassay	17	23		None			Lowe 1965[338]
96 hr static lab bioassay	13±1	26		95 percent confidence intervals			Earnest and Benville (unpublished)[334]
96 hr static lab bioassay	13±1	18		95 percent confidence intervals			Earnest and Benville ″
96 hr intermittent flow lab bioassay	14-18	28		95 percent confidence interval	0.13(0.02-0.27)		Earnest and Benville ″
96 hr intermittent flow lab bioassay	14-18	28		95 percent confidence intervals	0.11(0.08-0.15)		Earnest and Benville ″
96 hr static lab bioassay	13-18	12-30	Turb. 1-12 JTU	95 percent confidence intervals			Earnest (unpublished)[332]
96 hr static lab bioassay	20±.5	24	pH=8.0	None			Eisler 1969[327]
96 hr static lab bioassay	20±.5	24	pH=8.0	None			Eisler 1969[327]
96 hr static lab bioassay	20±.5	24	pH=8.0 DO 7.1-7.8	None			Eisler 1969[327]
96 hr static lab bioassay	20±.5	24	pH=8.0 DO 7.1-7.8	None			Eisler 1970a[328]
96 hr static lab bioassay	20±.5	24	pH=8.0 DO 7.1-7.8	None			Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24	pH=8.0 DO 7.1-7.8	None			Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24	pH=8.0 DO 7.1-7.8	None			Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24	pH=8.0 DO 7.1-7.8	None			Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24	pH=8.0 DO 7.1-7.8	None			Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24	pH=8.0 DO 7.1-7.8 (CaCo$_3$) 45-57 ppm	None			Eisler 1970b[329]

TABLE 6—

Substance Tested	Formulation	Organism Tested	Common Name	Life Stage or Size (mm)	Conc. (ppb act. ingred.) in water	Methods of Assessment
Heptachlor	100 percent	Sphaeroides maculatus	Northern puffer	168	188	LC-50
Heptachlor	72 percent	Gasterosteus aculeatus	Threespine stickleback	22–44	111.9	TLM
						Ratio O.D. Expt. Ratio O.D. Control
Lindane		Protococcus sp.			5,000	0.75 O.D. expt/O.D. control
Lindane		Chlorella sp.			5,000	0.57 O.D. exp/O.D. control
Lindane		Dunaliella euchlora			9,000	0.60 O.D. exp/O.D. control
Lindane		Phaeodactylum tricornutum			5,000	0.30 O.D. expt/O.D. control
Lindane		Monochrysis lutheri			5,000	1.00 O.D. expt O.D. control
Lindane	100 percent	Palaemon macrodactylus	Korean shrimp		12.5 (4.7–32.7)	TL-50
Lindane	100 percent	Palaemon macrodactylus	Korean shrimp		9.2 (5.8–15.0)	TL-50
Lindane	100 percent	Crangon septemspinosa	Sand shrimp	26	5	LC-50
Lindane	100 percent	Palaemonetes vulgaris	Grass shrimp	31	10.	LC-50
Lindane	100 percent	Pagurus longicarpus	Hermit crab	3.5	5.	LC-50
Lindane	100 percent	Nassa obsoleta	Mud snail	15	10000	Reduced deposition of egg cases from 1473 by control to 749 by Expt.
Lindane		Crassostrea virginica	Eastern oyster	Egg	9100	TLM
Lindane		Mercenaria mercenaria	Hard clam	Egg	>10000	TLM
Lindane		Mercenaria mercenaria	Hard clam	Larvae	>10000	TLM
Lindane	100 percent	Fundulus heteroclitus	Mummichog	42	20	LC-50
Lindane	100 percent	Fundulus heteroclitus	Mummichog	55	60	LC-50
Lindane	100 percent	Fundulus majalis	Striped killifish	49	28	LC-50
Lindane	100 percent	Menidia menidia	Atlantic silverside	57	9	LC-50
Lindane	100 percent	Mugil cephalus	Striped mullet	85	66	LC-50
Lindane	100 percent	Anguilla rostrata	American eel	56	56	LC-50
Lindane	100 percent	Thalossoma bifasciatum	Bluehead	90	14	LC-50
Lindane	100 percent	Sphaeroides maculatus	Northern puffer	168	35	LC-50
Lindane	100 percent	Gasterosteus aculeatus	Threespine stickleback	22–44	50	TLM
Methoxychlor	89.5 percent	Palaemon macrodactylus	Korean shrimp		.44 (0.21–0.93)	TL-50
Methoxychlor	89.5 percent	Palaemon macrodactylus	Korean shrimp		6.7 (4.37–10.7)	TL-50
Methoxychlor	100 percent	Crangon septemspinosa	Sand shrimp	26	4.	LC-50
Methoxychlor	100 percent	Palaemonetes vulgaris	Grass shrimp	31	12.	LC-50
Methoxychlor	100 percent	Pagurus longicarpus	Hermit crab	3.5	7.	LC-50
Methoxychlor	100 percent	Fundulus heteroclitus	Mummichog	42	35	LC-50
Methoxychlor	100 percent	Fundulus heteroclitus	Mummichog	65	35	LC-50
Methoxychlor	100 percent	Fundulus majalis	Striped killifish	40	30	LC-50
Methoxychlor	100 percent	Menidia menidia	Atlantic silverside	57	33	LC-50
Methoxychlor	100 percent	Mugil cephalus	Striped mullet	100	63	LC-50
Methoxychlor	100 percent	Anguilla rostrata	American eel	56	12.	LC-50
Methoxychlor	100 percent	Thalassoma bifasciatum	Bluehead	86	13.	LC-50
Methoxychlor	100 percent	Sphaeroides maculatus	Northern puffer	203	150.	LC-50
Methoxychlor	89.5 percent	Gasterosteus aculeatus	Threespine stickleback	22–44	69.1	TLM
Mirex	Technical	Tetrahymena pyriformis			0.9	16.03 percent decrease in population size
Mirex	Bait (.3 percent mirex)	Penaeus aztecus	Brown shrimp	24	One particle of mirex bait/shrimp	48 percent paralysis or death in 4 days
Mirex	Bait (.3 percent mirex)	Palaemonetes pugio	Grass shrimp	25	One particle of mirex bait/shrimp	63 percent paralysis/or death in 4 days
Mirex	Technical	Penaeus duorarum	Pink shrimp	55	1.0	100 percent paralysis/or death in 11 days
Mirex	Technical	Penaeus duorarum	Pink shrimp	55	0.1	36 percent paralysis/or death in 35 days
Mirex	Bait (0.3 percent mirex)	UCA pugilator	Fiddler crab	20	One particle of mirex bait per crab	73 percent paralysis/or death in 14 days
Mirex	Bait (0.3 percent) mirex	Callinectes sapidus	Blue crab	23	1 particle of bait/crab	84 percent paralysis/death in 20 days
Mirex		Callinectes sapidus	Blue crab	Adult	5.6×10^4 $(4.0\text{–}7.8)\times10^4$	TLM
TDE	99 percent	Palaemon macrodactylus	Korean shrimp		8.3 (4.8–14.4)	TL-50
TDE	99 percent	Palaemon macrodactylus	Korean shrimp		2.5 (1.6–4.0)	TL-50
Toxaphene	Polychloro dicyclic Terpenes with chlorated camphene 60 percent emulsion concentrate	Protococcus sp.			40	.77 O.D. expt/O.D. control
		Chlorella sp.			40	.70 O.D. expt/O.D. control
		Dunaliella euchlora			70	53 O.D. expt/O.D. control
		Phaeodactylum tricornutum			10	.54 O.D. expt/O.D. control
		Monochrysis lutheri			10	.00 O.D. expt/O.D. control
Toxaphene	100 percent	Palaemon macrodactylus	Korean shrimp		20.3 (8.6–47.9)	TL-50
Toxaphene		Callinectes sapidus	Blue crab	Adult	370 (180–700)	TLM
Toxaphene	Polychloro bicyclic Terpenes with chlorinated camphene Predominatory	Mercenaria mercenaria	Hard clam	Eggs	1120	TLM
		Mercenaria mercenaria	Hard clam	Larvae	250	TLM
Toxaphene	100 percent 67–69 percent CL	Gasterosteus aculeatus	Threespine stickleback	22–44	7.8	TLM
Thiodan®	96 percent	Palaemon macrodactylus	Korean shrimp		17.1 (8.4–39.8)	TL-50
Thiodan®	96 percent	Palaemon macrodactylus	Korean shrimp		3.4 (1.8–6.5)	TL-50

Continued

Test Procedure	Temp C	Salinity ‰	Other Environmental Criteria	Statistical Evaluation	Residue levels mg/kg	Other Parameters	Reference
96 hr static lab bioassay	20±.5	24	pH=8.0 DO 7.1-7.8	None	....	....	Eisler 1970b[329]
96 hr static lab bioassay	20±.5	25	pH 6 8-7.4 Total Alkalinity	None	....	....	Katz 1961[333]
Test organisms grown in test media containing pesticides for ten days Absorbance measured at 530 mμ	20±.5	22-28	500 ft.c-continuously	None	....	....	Ukeles 1962[347]
	20±.5	22-28	....	None	....	....	Ukeles 1962[347]
	20±.5	22-28	....	None	....	....	Ukeles 1962[347]
	20±.5	22-28	....	None	....	....	Ukeles 1962[347]
	20±.5	22-28	....	None	....	....	Ukeles 1962[347]
96 hr static lab bioassay	13-18	12-30	Turb 1-12 JTU	95 percent confidence intervals	....	....	Earnest (unpublished)[353]
96 hr flowing lab bioassay	13-18	12-30	....	95 percent confidence intervals	....	....	Earnest (unpublished)[353]
96 hr static lab bioassay	20±.5	24	pH=8.0 D.O. 7.1-7.7	None	....	....	Eisler 1969[327]
96 hr static lab bioassay	20±.5	24	pH=8.0 D O. 7.1-7.7	None	....	....	Eisler 1969[327]
96 hr static lab bioassay	20±.5	24	pH=8.0 D.O. 7.1-7.7	None	....	....	Eisler 1969[327]
96 hr static lab bioassay. Acute toxicity experiment followed by 133-day post exposure in clean water.	20±.5	24	pH=8.0	No. of eggs deposited significantly less at 0.001 level. $X^2>10.8$ Chi-square analysis	....	....	Eisler 1970c[330]
48 hr static lab bioassay	24±1	....	....	None	....	....	Davis and Hidu 1969[324]
48 hr static lab bioassay	24±1	....	....	None	....	....	Davis and Hidu 1969[324]
12 day static lab bioassay	24±1	....	....	None	....	....	Davis and Hidu 1969[324]
96 hr static lab bioassay	20±.5	24	pH=8.0 D.O. 7.1-7.7	None	....	....	Eisler 1970a[328]
96 hr static lab bioassay	20±.5	24	pH=8.0 D.O. 7.1-7.7	None	....	....	Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24	pH=8.0 D.O. 7.1-7.7	None	....	....	Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24	pH=8.0 O.D. 7.1-7.7	None	....	....	Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24	pH=8.0 O.D. 7.1-7.7	None	....	....	Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24	pH=8.0 O.D. 7.1-7.7	None	....	....	Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24	pH=8.0 O.D. 7.1-7.7	None	....	....	Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24	pH=8.0 O.D. 7.1-7.7	None	....	....	Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24	pH=8.0 O.D. 7.1-7.7	None	....	....	Katz 1961[333]
96 hr static lab bioassay	13-18	12-30	Turb. 1-12 JTU	95 percent confidence intervals	....	....	Earnest (unpublished)[353]
96 hr intermittent-flow lab bioassay	13-18	12-30	Turb. 1-12 JTU	95 percent confidence intervals	....	....	Earnest "
96 hr static lab bioassay	20±.5	24	pH=8.0 DO 7.1-7.7	None	....	....	Eisler 1969[327]
96 hr static lab bioassay	20±.5	24	pH=8.0 DO 7.1-7.7	None	....	....	Eisler 1969[327]
96 hr static lab bioassay	20±.5	24	pH=8.0 DO 7.1-7.7	None	....	....	Eisler 1969[327]
96 hr static lab bioassay	20	24	pH=8.0 DO 7.1-7.7	None	....	....	Eisler 1970a[328]
96 hr static lab bioassay	20±.5	24	pH=8.0 DO 7.1-7.7	None	....	....	Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24	pH=8.0 DO 7.1-7.7	None	....	....	Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24	pH=8.0 DO 7.1-7.7	None	....	....	Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24	pH=8.0 DO 7.1-7.7	None	....	....	Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24	pH=8.0 DO 7.1-7.7	None	....	....	Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24	pH=8.0 DO 7.1-7.7	None	....	....	Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24	pH=8.0 DO 7.1-7.7	None	....	....	Eisler 1970b[329]
96 hr static lab bioassay	20±.5	25	pH=6.8-7.4 Total alkalinity ($CaCo_3$) 45-57	None	....	....	Katz 1961[333]
96 hr growth test	26	0	Cultures grown in Tetrahymena broth	Measured effect is an average of the results of tests in which a significant difference existed ($P<0.05$)	....	....	Cooley et al. (unpublished)[351]
Static bioassay	22	21	....	....	....	....	Lowe et al. 1971a[340]
Static bioassay	25	33	....	....	1.1	....	Lowe et al. 1971a[340]
Flowing water bioassay	17	29 ‰	None	....	....	....	Lowe et al. 1971a[340]
Flowing water bioassay	14	29 ‰	None	....	0.26 ppm	....	Lowe et al. 1971a[340]
Flowing water bioassay	29	27 ‰	None	....	0.30 ppm	....	Lowe et al. 1971a[340]
96 hr flowing water bioassay	29	27	....	....	0.3	....	Lowe et al. 1971a[340]
96 hr static lab bioassay	21	19.3	....	95 percent confidence interval	....	....	Mahood et al. 1970[342]
96 hr static lab bioassay	13-18	12-30	Turb. 1-12 JTU	95 percent confidence interval	....	....	Earnest (unpublished)[353]
96 hr flowing water lab bioassay	13-18	12-30	Turb. 1-12 JTU	....	....	....	Earnest "
Test organisms grown in test media for 10 days absorbance measured at 530 m	20±.5	22-28	....	None	....	....	Ukeles 1962[347]
	20±.5	22-28	....	....	....	....	"
	20±.5	22-28	....	....	....	....	"
....	20±.5	22-28	....	....	....	....	"
....	20±.5	22-28	....	....	....	....	"
96 hr static lab bioassay	13-18	12-30	Turbidity 1-12 JTU	95 percent confidence interval	....	....	Earnest (unpublished)[353]
96 hr static lab bioassay	21	8.6	....	95 percent confidence interval	....	....	Mahood et al. 1970[342]
48 hr static lab bioassay	24±1	....	....	....	....	....	Davis and Hidu 1969[324]
12 day static lab bioassay	24±1	....	....	....	....	....	Davis and Hidu 1969[324]
96 hr static lab bioassay	20±.5	25	....	....	....	....	Katz 1961[333]
96 hr static lab bioassay	13-18	12-30	Turb 1-12 JTU	95 percent confidence interval	....	....	Earnest (unpublished)[353]
96 hr flowing water lab bioassay	13-18	12-30	Turb 1-12 JTU	95 percent confidence interval	....	....	Earnest "

TABLE 6—

Substance Tested	Formulation	Organism Tested	Common Name	Life Stage or Size (mm)	Conc. (ppb act. ingred.) in water	Methods of Assessment
Insecticides Organophosphates						
Abate		Dunaliella euchlora			1000	36 percent reduction in O_2 evolution
Abate		Dunaliella euchlora			100	23 percent reduction in O_2 evolution
Abate		Phaeodactylum tricornutum			1000	38 percent reduction in O_2 evolution
Abate		Phaeodactylum tricornutum			100	28 percent reduction in O_2 evolution
Abate		Skeletonema costatum			1000	55 percent reduction in O_2 evolution
Abate		Skeletonema costatum			100	23 percent reduction in O_2 evolution
Abate		Cyclotella nana			1000	80 percent reduction in O_2 evolution
Abate		Palaemon macrodactylus	Korean shrimp		2550 (994-6540)	TL-50
Abate		Palaemon macrodactylus	Korean shrimp		249 (72.5-853)	TL-50
Baytex		Dunaliella euchlora			1000.	27 percent reduction in O_2 evolution
Baytex		Dunaliella euchlora			100	27 percent reduction in O_2 evolution
Baytex		Dunaliella euchlora			10	16 percent reduction in O_2 evolution
Baytex		Phaeodactylum tricornutum			1000	29 percent reduction in O_2 evolution
Baytex		Phaeodactylum tricornutum			100	29 percent reduction in O_2 evolution
Baytex		Phaeodactylum tricornutum			10	35 percent reduction in O_2 evolution
Baytex		Skeletonema costatum			1000	19 percent reduction in O_2 evolution
Baytex		Skeletonema costatum			100	51 percent reduction in O_2 evolution
Baytex		Skeletonema costatum			10	26 percent recuction in O_2 evolution
Baytex		Cycictella nana			1000	50 percent reduction in O_2 evolution
Baytex		Cyclotella nana			100	48 percent reduction in O_2 evolution
Baytex		Palaemon macrodactylus	Korean shrimp		5.3 (3.13-8.92)	TL-50
Baytex		Palaemon macrodactylus	Korean shrimp		3.0 (1.5-6.0)	TL-50
CO-RAL		Crassostrea virginica	Eastern oyster	Egg	110	TLM
CO-RAL		Crassostrea virginica	Eastern oyster	Larvae	>1000	TLM
CO-RAL		Mercenaria mercenaria	Hard clam	Egg	9120	TLM
CO-RAL		Mercenaria mercenaria	Hard clam	Larvae	5210	TLM
CO-RAL		Gasterosteus aculeatus	Threespine stickleback	22–44	1470	TLM
DDVP		Crangon septemspinosa	Sand shrimp	26	4	LC-50
DDVP		Palaemonetes vulgaris	Grass shrimp	31	15	LC-50
DDVP		Pagurus longicarpus	Hermit crab	3.5	45	LC-50
DDVP		Fundulus heteroclitus	Mummichog	42	3700	LC-50
DDVP		Fundulus heteroclitus	Mummichog	55	2680	LC-50
DDVP		Fundulus majalis	Striped killifish	40	2300	LC-50
DDVP		Menidia menidia	Atlantic silverside	50	1250	LC-50
DDVP		Mugil cephalus	Striped mullet	84	200	LC-50
DDVP		Anguilla rostrata	American eel	59	1800	LC-50
DDVP		Thalassoma bifasciatum	Bluehead	80	1440	LC-50
DDVP		Sphaeroides maculatus	Northern puffer	168	2250	LC-50
Delnav	100 percent	Crangon septemspinosa	Sand shrimp	26	38	LC-50
Delnav	100 percent	Palaemonetes vulgaris	Grass shrimp	31	285	LC-50
Delnav	100 percent	Pagurus longicarpus	Hermit crab	3.5	82	LC-50
Dicapthon		Mercenaria mercenaria	Hard clam	Eggs	3340	TLM
Dicapthon		Mercenaria mercenaria	Hard clam	Larvae	5740	TLM
Dioxathion	100 percent	Fundulus heteroclitus	Mummichog	42	6	LC-50
Dioxathion	100 percent	Fundulus heteroclitus	Mummichog	56	20	LC-50
Dioxathion	100 percent	Fundulus majalis	Striped killifish	84	15	LC-50
Dioxathion	100 percent	Menidia menidia	Atlantic silverside	50	6	LC-50
Dioxathion	100 percent	Mugil cephalus	Striped mullet	85	39	LC-50
Dioxathion	100 percent	Anguilla rostrata	American eel	59	6	LC-50
Dioxathion	100 percent	Thalassoma bifasciatum	Bluehead	80	35	LC-50
Dioxathion	100 percent	Sphaeroides maculatus	Northern puffer	168	75	LC-50
Dipterex	50 percent soluble powder	Dunaliella euchlora			50,000	.54 (O.D. expt/O.D. cont.)
Dipterex	Soluble powder	Phaeodactylum tricornutum			50,000	.85 (O.D. expt/O.D. cont.)
Dipterex	Soluble powder	Phaeodactylum tricornutum			100,000	.39 (O D expt/O D. cont.)
Dipterex	Soluble powder	Protococcus sp.			100,000	.54 (O.D. expt/O.D. cont.)
Dipterex	Soluble powder	Chlorella sp.			50,000	.70 (O.D. expt/O.D. cont.)
Dipterex	Soluble powder	Chlorella sp.			500,000	.00 (O.D. expt/O.D. cont.)
Dipterex	Soluble powder	Monochrysis lutheri			50,000	.55 (O D. expt/O.D. cont.)
Dipterex		Crassostrea virginica	American oyster	Larvae	1,000	TLM
Di-syston		Crassostrea virginica	American oyster	Eggs	5860	TLM
Di-syston		Crassostrea virginica	American oyster	Larvae	3670	TLM
Di-syston		Mercenaria mercenaria	Hard clam	Eggs	55280	TLM
Di-syston		Mercenaria mercenaria	Hard clam	Larvae	1390	TLM
Dursban	Technical	Cymatogaster aggregata	Shiner perch	55	3.5	TLM
Dursban	Technical	Cymatogaster aggregata	Shiner perch	55	3.7	TLM
Dursban		Palaemon macrodactylus	Korean shrimp		0.25 (0.10-0.63)	TL-50
Dursban		Palaemon macrodactylus	Korean shrimp		0.01 (0.002-0.046)	TL-50
Guthion		Crassostrea virginica	American oyster	Eggs	620	TLM
Guthion		Mercenaria mercenaria	Hard clam	Eggs	860	TLM
Guthion		Mercenaria mercenaria	Hard clam	Larvae	860	TLM

Continued

Test Procedure	Temp C	Salinity ‰	Other Environmental Criteria	Statistical Evaluation		Residue levels mg/kg	Other Parameters	Reference
O_2 evolution measured by Winkler Light-and-Dark Bottle Technique 1 l. of culture incubated 20 hrs in pesticide soln. then placed in test bottles.			250 ft.-c for 4 hrs.	All percent significant at 0.05 level	t=6.1			Derby and Ruber 1971^{325}
					t=4.1			"
					t=3.8			"
					t=2.5			"
					t=4.8			"
					t=2.2			"
					t=6.8			"
96 hr static lab bioassay	13-18	12-30	Turb. 1-12 JTU	95 percent confidence intervals				Earnest (unpublished)^{353}
96 hr intermittent flow lab bioassay	13-18	12-30		95 percent confidence intervals				Earnest (unpublished)^{353}
O_2 evolution measured by Winkler Light-and-Dark Bottle Technique			250 ft.-c for 4 hrs	All percent significant at 0.05 level	t=5.4			Derby and Ruber 1971^{325}
					t=6.7			"
					t=2.6			"
					t=2.5			"
1 l. of culture incubated 20 hrs in pesticide soln. then placed in test bottles.					t=2.5			"
					t=3.5			"
					t=2.3			"
					t=5.9			"
					t=3.2			"
					t=3.8			"
					t=2.7			"
96 hr static lab bioassay	13-18	12-30	Turb. 1-12 JTU	95 percent confidence intervals				Earnest (unpublished)^{353}
96 hr intermittent flow lab bioassay	13-18	12-30	Turb. 1-12 JTU	95 percent confidence intervals				"
48 hr static lab bioassay	24±1	22-28		None				Davis and Hidu 1969^{324}
14 day static lab bioassay	24±1	22-28		None				"
48 hr static lab bioassay	24±1	22-28		None				"
12 day static lab bioassay	24±1	22-28		None				"
96 hr static lab bioassay	20±.5	25	pH 6.8-7.4 total alkalinity 45-57 ppm	None				Katz 1961^{333}
96 hr static lab bioassay	20±.5	24	pH 8.0 DO 7.1-7.7	None				Eisler 1969^{327}
96 hr static lab bioassay	20±.5	24	pH 8.0 DO 7.1-7.7	None				Eisler 1969^{327}
96 hr static lab bioassay	20±.5	24	pH 8.0 DO 7.1-7.7	None				Eisler 1969^{327}
96 hr static lab bioassay	20	24	pH 8.0 DO 7.0-7.7	None				Eisler 1970a^{328}
96 hr static lab bioassay	20±.5	24	pH 8.0 DO 7.1-7.7	None				Eisler 1970b^{329}
96 hr static lab bioassay	20±.5	24	pH 8.0 DO 7.1-7.7	None				Eisler 1970b^{329}
96 hr static lab bioassay	20±.5	24	pH 8.0 DO 7.1-7.7	None				Eisler 1970b^{329}
96 hr static lab bioassay	20±.5	24	pH 8.0 DO 7.1-7.7	None				Eisler 1970b^{329}
96 hr static lab bioassay	20±.5	24	pH 8.0 DO 7.1-7.7	None				Eisler 1970b^{329}
96 hr static lab bioassay	20±.5	24	pH 8.0 DO 7.1-7.7	None				Eisler 1970b^{329}
96 hr static lab bioassay	20±.5	24	pH 8.0 DO 7.1-7.7	None				Eisler 1970b^{329}
96 hr static lab bioassay	20±.5	24	pH 8.0 DO 7.1-7.7	None				Eisler 1969^{327}
96 hr static lab bioassay	20±.5	24	pH 8.0 DO 7.1-7.7	None				Eisler 1969^{327}
96 hr static lab bioassay	20±.5	24	pH 8.0 DO 7.1-7.7	None				Eisler 1969^{327}
48 hr static lab bioassay	24±1.	22-28		None				Davis and Hidu 1969^{324}
12 day static lab bioassay	24±1.	22-28		None				Davis and Hidu 1969^{324}
96 hr static lab bioassay	20	24	pH 8.0 DO 7.0-7.7	None				Eisler 1970a^{328}
96 hr static lab bioassay	20±.5	24	pH 8.0	None				Eisler 1970b^{329}
96 hr static lab bioassay	20±.5	24	pH 8.0	None				Eisler 1970b^{329}
96 hr static lab bioassay	20±.5	24	pH 8.0	None				Eisler 1970b^{329}
96 hr static lab bioassay	20±.5	24	pH 8.0	None				Eisler 1970b^{329}
96 hr static lab bioassay	20±.5	24	pH 8.0	None				Eisler 1970b^{329}
96 hr static lab bioassay	20±.5	24	pH 8.0	None				Eisler 1970b^{329}
96 hr static lab bioassay	20±.5	24	pH 8.0	None				Eisler 1970b^{329}
Organisms grown in test media containing pesticide for 10 days optical density measured at 530 mμ	20.5±1	22-28		None				Ukeles 1962^{347}
	20.5±1	22-28		None				Ukeles 1962^{347}
	20.5±1	22-28		None				Ukeles 1962^{347}
	20.5±1	22-28		None				Ukeles 1962^{347}
	20.5±1	22-28		None				Ukeles 1962^{347}
	20.5±1	22-28		None				Ukeles 1962^{347}
		22-28		None				Ukeles 1962^{347}
48 hr static lab bioassay	24±1.			None				Davis and Hidu 1969^{324}
48 hr static lab bioassay	24±1.			None				Davis and Hidu 1969^{324}
14 day static lab bioassay	24±1.			None				"
48 hr static lab bioassay	24±1.			None				"
12 day static lab bioassay	24±1.			None				"
96 hr static lab bioassay	20.5±1.	25		None				Millemann 1969^{343}
96 hr flowing water lab bio.	20.5±1.	25		None				Millemann 1969^{343}
96 hr static lab bioassay	13-18	12-30	Turb. 1-12 JTU	95 percent confidence interval				Earnest (unpublished)^{353}
96 hr. intermittent flow lab bioassay	13-18	12-30	Turb. 1-12 JTU	95 percent confidence interval				Earnest "
48 hr static lab bioassay	24±1.			None				Davis and Hidu 1969^{324}
48 hr static lab bioassay	24±1.			None				Davis and Hidu 1969^{324}
12 day static lab bioassay	24±1.			None				Davis and Hidu 1969^{324}

TABLE 6—

Substance Tested	Formulation	Organism Tested	Common Name	Life Stage or Size (mm)	Conc. (ppb act. ingred.) in water	Methods of Assessment
Guthion		Cyprinodon variegatus	Sheepshead minnow	40–70	2	Acetylcholinesterase activity[d] in control-vs-experimental groups control=1.36 expt.=0.097
Guthion	93 percent	Gasterosteus aculeatus	Threespine stickleback	22–44	4.8	TLM
Malathion		Tetrahymena pyriformis		Log-phase	10,000	8.8 percent decrease in a population size measured as absorbance at 540 $m\mu$
Malathion	95 percent	Palaemon macrodactylus	Korean shrimp		81.5 (19.6–26.1)	TL-50
Malathion	95 percent	Palaemon macrodactylus	Korean shrimp		33.7 (21.3–53.1)	TL-50
Malathion	100 percent	Crangon septemspinosa	Sand shrimp	26	33	LC-50
Malathion	100 percent	Palaemonetes vulgaris	Grass shrimp	31	82	LC-50
Malathion	100 percent	Pagurus longicarpus	Hermit crab	35	83	LC-50
Malathion		Crassostrea virginica	American oyster	Egg	9070	TLM
Malathion		Crassostrea virginica	American oyster	Larvae	2660	TLM
Malathion	100 percent	Fundulus heteroclitus	Mummichog	42	70	LC-50
Malathion	100 percent	Fundulus heteroclitus	Mummichog	56	80	LC-50
Malathion	100 percent	Fundulus majalis	Striped killifish	84	250	LC-50
Malathion	100 percent	Menidia menidia	Atlantic silverside	50	125	LC-50
Malathion	100 percent	Mugil cephalus	Striped mullet	48	550	LC-50
Malathion	100 percent	Thalassoma bifasciatum	Bluehead	80	27	LC-50
Malathion	100 percent	Anguilla rostrata	American eel	57	82	LC-50
Malathion	100 percent	Sphaeroides maculatus	Northern puffer	183	3250	LC-50
Malathion	57 percent	Gasterosteus aculeatus	Threespine stickleback	22–44	76.9	TLM
Methyl Parathion	100 percent	Crangon septemspinosa	Sand shrimp	26	2	LC-50
Methyl Parathion	100 percent	Palaemonetes vulgaris	Grass shrimp	31	3	LC-50
Methyl Parathion	100 percent	Pagurus longicarpus	Hermit crab	3.5	7	LC-50
Methyl Parathion		Fundulus heteroclitus	Mummichog	38	8,000	LC-50
Methyl Parathion	100	Fundulus heteroclitus	Mummichog	55	58,000	LC-50
Methyl Parathion	100	Fundulus majalis	Striped killifish	84	13,800	LC-50
Methyl Parathion	100	Menidia menidia	Atlantic silverside	50	5,700	LC-50
Methyl Parathion	100	Mugil cephalus	Striped mullet	48	5,200	LC-50
Methyl Parathion	100	Anguilla rostrata	American eel	59	16,900	LC-50
Methyl Parathion	100	Thalassoma bifasciatum	Bluehead	90	12,300	LC-50
Methyl Parathion	100	Sphaeroides maculatus	Northern puffer	196	75,800	LC-50
Parathion		Cyprinodon variegatus	Sheepshead minnow	40–70	10	Acetylcholinesterase activity[d] in control-vs.-expt. groups Control=1.36 Expt.=0.120
Phorate		Cyprinodon variegatus	Sheepshead minnow	40–70	5	Acetylcholinesterase activity[d] in control-vs-expt groups Control=1.36 Expt=0.086
Phosdrin®	100 percent	Crangon septemspinosa	Sand shrimp	26	11	LC-50
Phosdrin®	100 percent	Palaemonetes vulgaris	Grass shrimp	31	69	LC-50
Phosdrin®	100 percent	Pagurus longicarpus	Hermit crab	3.5	28	LC-50
Phosdrin®	100 percent	Fundulus heteroclitus	Mummichog	42	65	LC-50
Phosdrin®	100 percent	Fundulus heteroclitus	Mummichog	56	300	LC-50
Phosdrin®	100 percent	Fundulus majalis	Striped killifish	84	75	LC-50
Phosdrin®	100 percent	Menidia menidia	Atlantic silverside	50	320	LC-50
Phosdrin®	100 percent	Mugil cephalus	Striped mullet	100	300	LC-50
Phosdrin®	100 percent	Anguilla rostrata	American eel	59	65	LC-50
Phosdrin®	100 percent	Thalassoma bifasciatum	Bluehead	80	74	LC-50
Phosdrin®	100 percent	Sphaeroides maculatus	Northern puffer	168	800	LC-50
TEPP		Protococcus sp.			1×10^5	.62 OD expt/OD control
TEPP		Protococcus sp.			5×10^5	.00 OD expt/OD control
TEPP		Chlorella sp.			1×10^5	.65 OD expt/OD control
TEPP		Chlorella sp.			3×10^5	.27 OD expt/OD control
TEPP		Dunaliella euchlora			3×10^5	.49 OD expt/OD control
TEPP		Phaeodactylum tricornutum			1×10^5	.58 OD expt/OD control
TEPP		Monochrysis lutheri			1×10^5	.83 OD expt/OD control
TEPP		Monochrysis lutheri			3×10^5	.38 OD expt/OD control
TEPP		Crassostrea virginica	American oyster	Egg	$>1\times10^4$	TLM
TEPP		Crassostrea virginica	American oyster	Larvae	$>1\times10^4$	TLM
Insecticides Carbamates						
Baygon		Dunaliella euchlora			1000	25 percent reduction in O_2 evolution
Baygon		Dunaliella euchlora			100	32 percent reduction in O_2 evolution
Baygon		Dunaliella euchlora			10	27 percent reduction in O_2 evolution
Baygon		Phaeodactylum tricornutum			1000	23 percent reduction in O_2 evolution
Baygon		Phaeodactylum tricornutum			100	28 percent reduction in O_2 evolution
Baygon		Phaeodactylum tricornutum			10	40 percent reduction in O_2 evolution
Baygon		Skeletonema costatum			1000	30 percent reduction in O_2 evolution
Baygon		Skeletonema costatum			100	23 percent reduction in O_2 evolution
Baygon		Skeletonema costatum			10	29 percent reduction in O_2 evolution
Baygon		Cyclotella nana			1000	53 percent reduction in O_2 evolution

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[d] ACh hydrolysed/hr/mg. brain

Continued

Test Procedure	Temp C	Salinity ‰	Other Environmental Criteria	Statistical Evaluation	Residue levels mg/kg	Other Parameters	Reference
72 hr static exposure	21±2.	4	pH=7.±0.2	Statistical difference at 0.001 level t=21.40	...	...	Coppage (unpublished)[352]
96 hr static lab bioassay	.	25	pH 6.8-7.4 Total Alkalinity as CaCo 45-57	None	...	...	Katz 1961[333]
96 hr growth test in Tetrahymena broth	26	0		Statistical difference at 0.05 level	...	...	Cooley and Keltner (unpublished)[350]
96 hr static lab bioassay	13-18	12-30	Turb 1-12 JTU	95 percent confidence interval	...	...	Earnest (unpublished)[353]
96 hr intermittent flow lab bioassay	13-18	12-30	Turb 1-12 JTU	95 percent confidence interval	...	...	Earnest (unpublished)[353]
96 hr static lab bioassay	20±.5	24	pH=8.0 DO 7.1-7.8	None	...	...	Eisler 1969[327]
96 hr static lab bioassay	20±.5	24	pH=8.0 DO 7.1-7.8	None	...	...	Eisler 1969[327]
96 hr static lab bioassay	20±.5	24	pH=8.0 DO 7.1-7.8	None	...	...	Eisler 1969[327]
48 hr static lab bioassay	24±1.	...	...	None	...	...	Davis and Hidu 1969[324]
14 day static lab bioassay	24±1.	...	...	None	...	...	"
96 hr static lab bioassay	20	24	pH=8.0 DO 7.0-7.7	None	...	...	Eisler 1970a[328]
96 hr static lab bioassay	20±.5	24	pH=8.0	None	...	...	Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24	pH=8.0	None	...	...	Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24	pH=8.0	None	...	...	Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24	pH=8.0	None	...	...	Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24	pH=8.0	None	...	...	Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24	pH=8.0	None	...	...	Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24	pH=8.0	None	...	...	Eisler 1970b[329]
96 hr static lab bioassay	20±.5	25	pH=6.8-7.4 45-57 Total alkalinity as CaCo	None	...	...	Katz 1961[332]
96 hr static lab bioassay	20±.5	24	pH=8.0 DO 7.1-7.7	None	...	...	Eisler 1969[327]
96 hr static lab bioassay	20±.5	24	pH=8.0 DO 7.1-7.7	None	...	...	Eisler 1969[327]
96 hr static lab bioassay	20±.5	24	pH=8.0 DO 7.1-7.7	None	...	...	"
96 hr static lab bioassay	20±.5	24	pH=8.0 DO 7.0-7.7	None	...	...	Eisler 1970a[328]
96 hr static lab bioassay	20±.5	24	pH=8.0	None	...	...	Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24		None	...	...	Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24		None	...	...	Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24		None	...	...	Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24		None	...	...	Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24		None	...	...	Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24		None	...	...	Eisler 1970b[329]
72 hr static exposure	21±2.	4	pH 7±.2	Statistically different at 0.001 level t=21.0169	...	...	Coppage (unpublished)[352]
72 hr static exposure	21±2.	4	pH 7±.2	Statistically different at 0.001 level t=4.603	...	...	Coppage (unpublished)[352]
96 hr static lab bioassay	20±.5	24	pH=8.0 DO=7.1-7.7	None	...	...	Eisler 1969[327]
96 hr static lab bioassay	20±.5	24	pH=8.0 DO=7.1-7.7	None	...	...	Eisler 1969[327]
96 hr static lab bioassay	20±.5	24	pH=8.0 DO=7.1-7.7	None	...	...	Eisler 1969[327]
96 hr static lab bioassay	20	24	pH=8.0	None	...	...	Eisler 1970a[328]
96 hr static lab bioassay	20±.5	24	pH=8.0 DO=7.1-7.7	None	...	...	Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24	pH=8.0 DO=7.1-7.7	None	...	...	Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24	pH=8.0 DO=7.1-7.7	None	...	...	Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24	pH=8.0 DO=7.1-7.7	None	...	...	Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24	pH=8.0 DO=7.1-7.7	None	...	...	Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24	pH=8.0 DO=7.1-7.7	None	...	...	Eisler 1970b[329]
96 hr static lab bioassay	20±.5	24	pH=8.0 DO=7.1-7.7	None	...	...	Eisler 1970b[329]
10 day growth test	20.5±1	22-28	500 ft.-c continuous	None	...	...	Ukeles 1962[347]
10 day growth test	20.5±1	22-28	500 ft.-c continuous	None	...	...	Ukeles 1962[347]
10 day growth test	20.5±1	22-28	500 ft.-c continuous	None	...	...	Ukeles 1962[347]
10 day growth test	20.5±1	22-28	500 ft.-c continuous	None	...	...	Ukeles 1962[347]
10 day growth test	20.5±1	22-28	500 ft.-c continuous	None	...	...	Ukeles 1962[347]
10 day growth test	20.5±1	22-28	500 ft.-c continuous	None	...	...	Ukeles 1962[347]
10 day growth test	20.5±1	22-28	500 ft.-c continuous	None	...	...	Ukeles 1962[347]
10 day growth test	20.5±1	22-28	500 ft.-c continuous	None	...	...	Ukeles 1962[347]
48 hr static lab bioassay	24±1	...	...	None	...	...	Davis and Hidu 1969[324]
14 day static lab bioassay	24±1	...	...	None	...	...	Davis and Hidu 1969[324]
O_2 evolution measured by Winkler Light-and-Dark Bottle technique 1 l. of culture incubated 20 hrs in pesticide solution, then placed in test bottles 4 hrs.	...	...	250 ft.-c 4 hrs	All percent significant at 0.05 level t=4.5	...	...	Derby and Ruber 1971[325]
	...	...	...	t=4.6	...	...	Derby and Ruber 1971[325]
	...	...	...	t=6.8	...	...	Derby and Ruber 1971[325]
	...	...	...	t=1.8	...	...	Derby and Ruber 1971[325]
	...	...	...	t=2.5	...	...	Derby and Ruber 1971[325]
...	...	...	...	t=3.8	...	...	Derby and Ruber 1971[325]
...	...	...	...	t=4.3	...	...	Derby and Ruber 1971[325]
...	...	...	...	t=2.1	...	...	Derby and Ruber 1971[325]
...	...	...	...	t=2.9	...	...	Derby and Ruber 1971[325]
...	...	...	...	t=11.0	...	...	Derby and Ruber 1971[325]

TABLE 6—

Substance Tested	Formulation	Organism Tested	Common Name	Life Stage or Size (mm)	Conc. (ppb act. ingred.) in water	Methods of Assessment
Sevin®	95 percent	Dunaliella euchlora			1000	.65 O.D. expt/O.D. control
Sevin®	95 percent	Phaeodactylum tricornutum			100	[*] .00 O.D. expt/O.D. control
Sevin®	95 percent	Monochrysis lutheri			1000	[*] .00 O.D. expt/O.D. control
Sevin®	95 percent	Chlorella sp.			1000	.80 O.D. expt/O.D. control
Sevin®	95 percent	Chlorella sp.			10,000	.00 O.D. expt/O.D. control
Sevin®	95 percent	Protococcus sp.			1000	.74 O.D. expt/O.D. control
Sevin®	95 percent	Protococcus sp.			10,000	[*] .00 O.D. expt/O.D. control
Sevin®	100 percent	Palaemon macrodactylus	Korean shrimp		12.0 (8.5–13.5)	TL-50
Sevin®	100 percent	Palaemon macrodactylus	Korean shrimp		7.0 (1.5–28)	TL-50
Sevin®	80 percent	Upogebia pugettensis	Mud shrimp	3	40 (30–60)	TLM
Sevin®	80 percent	Callianassa californiensis	Ghost shrimp	3	30	TLM
Sevin®	80 percent	Callianassa californiensis	Ghost shrimp	Adult	130	TLM
Sevin®	80 percent	Cancer magister	Dungeness crab	Juvenile (male)	600 (590–610)	EC-50 (Paralysis or death) loss of equilibrium
Sevin®	80 percent	Hemigrapsis oregonensis	Shore crab	Adult (female)	270 (60–690)	EC-50 (Paralysis loss of equilibrium or death)
Sevin®	80 percent	Crassostrea gigas	Pacific oyster	Larvae	2200 (1500–2700)	EC-50 prevention of development to straight linge shell stage.
Sevin®		Crassostrea virginica	American oyster	Eggs	3,000	TLM
Sevin®		Crassostrea virginica	American oyster	Larvae	3,000	TLM
Sevin®		Mercanaria mercenaria	Hard clam	Eggs	3,820	TLM
Sevin®		Mercenaria mercenaria	Hard clam	Larvae	>2,500	TLM
Sevin®	80 percent	Clinocardium nuttall	Cockle clam	Adults	7,300	TLM
Sevin®	80 percent	Clinocardium nuttalli	Cockle clam	Juvenile	3,850	TLM
Sevin®	80 percent	Mytilus edulis	Bay mussel	Larvae	2,300 (1400–2900)	EC-50 prevention of development to straight linge shell stage.
Sevin®	80 percent	Parophrys vetulus	English sole	Juvenile	4,100 (3200–5000)	TLM
Sevin®	80 percent	Cymatogaster aggregata	Shiner perch	Juvenile	3,900 (3800–4000)	TLM
Sevin®	80 percent	Gasterosteus aculeatus	Threespine stickleback	Juvenile	6,700 (5500–7700)	TLM
Sevin®	95 percent	Gasterosteus aculeatus	Threespine stickleback	22–44	3,990	TLM
Sevin®	98 percent	Leiostomus xanthurus	Spot	18 mm	100	65 percent survived in experimental and control test
Sevin®	80 percent	Onchorynchus keta	Chum salmon	Juvenile	2,500 (2200–2700)	TLM
Sevin®	80 percent	Cancer magister	Dungeness crab	egg/prezoeal	6	Prevention of hatching and molting
Sevin®	80 percent	Cancer magister	Dungeness crab	Zoea	10	Prevention of molting and death
Sevin®	80 percent	Cancer magister	Dungeness crab	Juvenile	280	Death or paralysis
Sevin®	80 percent	Cancer magister	Dungeness crab	Adult	180	Death or paralysis
Insecticides Miscellaneous						
Apholate		Palaemonetes vulgaris	Grass shrimp	29.5	$>5\times10^6$	TLM
Apholate		Palaemonetes vulgaris	Grass shrimp	29.5	5.50×10^5	Post exposure TLM
Apholate		Nassa obsoleta	Mud snail	13.4	$>3\times10^6$	TLM
Apholate		Nassa obsoleta	Mud snail	13.76	1.0×10^4	Reduction in the # of egg cases deposited from 103 for control to 70 for expt.
Apholate		Nassa obsoleta	Mud snail	12.71	1.0×10^5	Reduction in # of egg cases deposited from 103 by control to 16 by expt.
Apholate		Fundulus majalis	Striped killifish	41.5	$>5.\times10^6$	TLM
Herbicides Benzoic acid						
Chloramben	Technical acid	Chlorococcum sp.			1.15×10^5	50 percent decrease in O_2 evolution
"	Technical acid	Chlorococcum sp.			$5.\times10^4$	50 percent decrease in growth
"	Technical acid	Dunaliella tertiolecta			1.5×10^5	50 percent decrease in O_2 evolution
"	Technical acid	Dunaliella tertiolecta			$5.\times10^4$	50 percent decrease in growth
"	Technical acid	Isochrysis galbana			1×10^5	50 percent decrease in O_2 evolution
"	Technical acid	Isochrysis galbana			1.5×10^4	50 percent decrease in growth
"	Technical acid	Phaeodactylum tricornutum			1.0×10^5	50 percent decrease in O_2 evolution
"	Technical acid	Phaeodactylum tricornutum			2.5×10^4	50 percent decrease in growth
"	Ammonium salt	Chlorococcum sp.			2.225×10^6	50 percent decrease in O_2 evolution
"	Ammonium salt	Chlorococcum sp.			$4.\times10^6$	50 percent decrease in growth
"	Ammonium salt	Dunaliella tertiolecta			2.75×10^6	50 percent decrease in O_2 evolution
"	Ammonium salt	Dunaliella tertiolecta			$4.\times10^6$	50 percent decrease in growth
"	Ammonium salt	Isochrysis galbana			1.5×10^6	50 percent decrease in O_2 evolution
"	Ammonium salt	Isochrysis galbana			3.5×10^6	50 percent decrease in growth

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[*] No growth but organisms were viable.

Continued

Test Procedure	Temp C	Salinity ‰	Other Environmental Criteria	Statistical Evaluation	Residue levels mg/kg	Other Parameters	Reference
10 day growth test	20.5±1	22–28	500 ft.-c continuous	None	......	......	Ukeles 1962[347]
10 day growth test	20.5±1	22–28	500 ft.-c continuous	None	......	......	Ukeles 1962[347]
10 day growth test	20.5±1	22–28	500 ft.-c continuous	None	......	......	Ukeles 1962[347]
10 day growth test	20.5±1	22–28	500 ft.-c continuous	None	......	......	Ukeles 1962[347]
10 day growth test	20.5±1	22–28	500 ft.-c continuous	None	......	......	Ukeles 1962[347]
10 day growth test	20.5±1	22–28	500 ft.-c continuous	None	......	......	Ukeles 1962[347]
10 day growth test	20.5±1	22–28	500 ft.-c continuous	None	......	......	Ukeles 1962[347]
96 hr static lab bioassay	13–18	12–30	Turbidity 1–12 JTU	95 percent confidence limits	......	......	Earnest (unpublished)[843]
96 hr intermittent-flow lab bioassay	13–18	12–30	Turbidity 1–12 JTU	95 percent confidence limits	......	......	Earnest (unpublished)[843]
48 hr static lab bioassay	20±2	25	pH 7.9–8.1	None	......	......	Stewart et al. 1967[346]
48 hr static lab bioassay	20±2	25	pH 7.9–8.1	None	......	......	Stewart et al. 1967[346]
24 hr static lab bioassay	20±2	25	pH 7.9–8.1	None	......	......	Stewart et al. 1967[346]
24 hr static lab bioassay	20±2	25	pH 7.9–8.1	None	......	......	Stewart et al. 1967[346]
24 hr static lab bioassay	20±2	25	pH 7.9–8.1	None	......	......	Stewart et al. 1967[346]
48 hr static lab bioassay	20±2	25	pH 7.9–8.1	None	......	......	Stewart et al. 1967[346]
48 hr static lab bioassay	24±1	......	......	None	......	......	Davis and Hidu 1969[234]
14 day static lab bioassay	24±1	......	......	None	......	......	Davis and Hidu 1969[234]
14 hr static lab bioassay	24±1	......	......	None	......	......	Davis and Hidu 1969[234]
14 day static lab bioassay	24±1	......	......	None	......	......	Davis and Hidu 1969[234]
24 hr static lab bioassay	20±2	25	pH 7.9–8.1	None	......	......	Stewart et al. 1967[346]
96 hr static lab bioassay	20±1	25	......	None	......	......	Butler et al. 1968[222]
96 hr static lab bioassay	20±2	25	pH 7.9–8.1	None	......	......	Stewart et al. 1967[346]
24 hr static lab bioassay	20±2	25	pH 7.9–8.1	None	......	......	Stewart et al. 1967[346]
24 hr static lab bioassay	20±2	25	pH 7.9–8.1	None	......	......	Stewart et al. 1967[346]
24 hr static lab bioassay	20±2	25	pH 7.9–8.1	None	......	......	Stewart et al. 1967[346]
96 hr static lab bioassay	20±.5	25	pH=6.8–7.4 Total alkalinity 45–75 ppm	None	......	......	Katz 1961[333]
5 months continuous flow chronic lab bioassay	16–29	24–30	......	None	......	No pathology; mild AChE inhibition.	Lowe 1967[339]
96 hr static lab bioassay	15	25	......	None	......	......	Millemann 1969[343]
24 hr static lab bioassay	10±1	25	......	None	......	......	Buchanan et al. 1969[221]
96 hr static lab bioassay	10±1	25	......	None	......	......	Buchanan et al. 1969[221]
96 hr static lab bioassay	18±1	25	......	None	......	......	Buchanan et al. 1969[221]
96 hr static lab bioassay	18	25	......	None	......	......	Buchanan et al. 1969[221]
96 hr static lab bioassay	20	24	pH 7.8	None	......	......	Eisler 1966[726]
50 days static conditions	20	24	pH 7.8	None	......	......	Eisler 1966[726]
96 hr static lab bioassay	20	24	pH 7.8	None	......	......	Eisler 1966[726]
100 day post exposure to 96 hr static lab bioassay at 10 ppm.	20	24	pH 7.8	Reduction significant at 0.01 level. Analysis by Chi-square	......	......	Eisler 1966[726]
	20	24	pH 7.8	Reduction significant at 0.01 level. Analysis by Chi-square	......	......	Eisler 1966[726]
96 hr static lab bioassay	20	24	pH 7.8	None	......	......	Eisler 1966[726]
f	20	30	pH=7.9–8.1 6000 lux 12/12	Litchfield & Wilcoxon Method, 1947[337]	......	......	Walsh 1972[318]
Growth measured as ABS. (525 mu) after 10 days	20	30	pH=7.9–8.1 6000 lux 12/12	"	......	......	Walsh "
f	20	30	pH=7.9–8.1 6000 lux 12/12	"	......	......	Walsh "
Growth measured as ABS. (525 mu) after 10 days	20	30	pH=7.9–8.1 6000 lux 12/12	"	......	......	Walsh "
f	20	30	pH=7.9–8.1 6000 lux 12/12	"	......	......	Walsh "
Growth measured as ABS. (525 mu) after 10 days	20	30	pH=7.9–8.1 6000 lux 12/12	"	......	......	Walsh "
f	20	30	pH=7.9–8.1 6000 lux 12/12	"	......	......	Walsh "
Growth measured as ABS. (525 mu) after 10 days	20	30	pH=7.9–8.1 6000 lux 12/12	"	......	......	Walsh "
f	20	30	pH=7.9–8.1 6000 lux 12/12	"	......	......	Walsh "
Growth measured as ABS. (525 mu) after 10 days	20	30	pH=7.9–8.1 6000 lux 12/12	"	......	......	Walsh "
f	20	30	pH=7.9–8.1 6000 lux 12/12	"	......	......	Walsh "
Growth measured as ABS. (525 mu) after 10 days	20	30	pH=7.9–8.1 6000 lux 12/12	"	......	......	Walsh "
f	20	30	pH=7.9–8.1 6000 lux 12/12	"	......	......	Walsh "
Growth measured as ABS. (525 mu) after 10 days	20	30	pH=7.9–8.1 6000 lux 12/12	"	......	......	Walsh "

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f O_2 evolution measured by Gilson differential respirometer on 4 ml of culture in log phase. Length of test 90 minutes.

TABLE 6—

Substance Tested	Formulation	Organism Tested	Common Name	Life Stage or Size (mm)	Conc. (ppb act. ingred.) in water	Methods of Assessment
"	Ammonium salt	Phaeodactylum tricornutum			3.25×10^{6}	50 percent decrease in O_2 evolution
"	Ammonium salt	Phaeodactylum tricornutum			3.0×10^{5}	50 percent decrease in growth
"	Methyl ester	Chlorococcum sp.			2×10^{3}	50 percent decrease in O_2 evolution
"	Methyl ester	Chlorococcum sp.			2.5×10^{3}	50 percent decrease in growth
"	Methyl ester	Dunaliella tertiolecta			1.75×10^{3}	50 percent decrease in O_2 evolution
"	Methyl ester	Dunaliella tertiolecta			5×10^{3}	50 percent decrease in growth
"	Methyl ester	Isochrysis galbana			1.5×10^{3}	50 percent decrease in O_2 evolution
"	Methyl ester	Isochrysis galbana			5×10^{3}	50 percent decrease in growth
"	Methyl ester	Phaeodactylum tricornutum			2.75×10^{3}	50 percent decrease in O_2 evolution
"	Methyl ester	Phaeodactylum tricornutum			$5.\times10^{3}$	50 percent decrease in growth
Dipyridylium						
Diquat	Dibromide	Chlorococcum sp.			$>.5\times10^{5}$	50 percent decrease in O_2 evolution
Diquat	Dibromide	Chlorococcum sp.			$2.\times10^{5}$	50 percent decrease in growth
Diquat	Dibromide	Dunaliella tertiolecta			$>5.\times10^{5}$	50 percent decrease in O_2 evolution
Diquat	Dibromide	Dunaliella tertiolecta			3×10^{4}	50 percent decrease in growth
Diquat	Dibromide	Isochrysis galbana			$>5\times10^{5}$	50 percent decrease in O_2 evolution
Diquat	Dibromide	Isochrysis galbana			1.5×10^{4}	50 percent decrease in growth
Diquat	Dibromide	Phaeodactylum tricornutum			$>5\times10^{5}$	50 percent decrease in O_2 evolution
Diquat	Dibromide	Phaeodactylum tricornutum			1.5×10^{4}	50 percent decrease in growth
Paraquat	Dichloride	Chlorococcum sp.			$>5\times10^{5}$	50 percent decrease in O_2 evolution
Paraquat	Dichloride	Chlorococcum sp.			8×10^{4}	50 percent decrease in growth
Paraquat	Dichloride	Dunaliella tertiolecta			2.5×10^{5}	50 percent decrease in O_2 evolution
Paraquat	Dichloride	Dunaliella tertiolecta			2×10^{4}	50 percent decrease in growth
Paraquat	Dichloride	Isochrysis galbana			5×10^{5}	50 percent decrease in O_2 evolution
Paraquat	Dichloride	Isochrysis galbana			5×10^{3}	50 percent decrease in growth
Paraquat	Dichloride	Phaeodactylum tricornutum			3.5×10^{4}	50 percent decrease in O_2 evolution
Paraquat	Dichloride	Phaeodactylum tricornutum			$1.\times10^{4}$	50 percent decrease in growth
Nitrile						
Dichlobenil	Technical acid	Chlorococcum sp.			9×10^{4}	50 percent decrease in O_2 evolution
Dichlobenil	Technical acid	Chlorococcum sp.			6×10^{4}	50 percent decrease in growth
Dichlobenil	Technical acid	Dunaliella tertiolecta			1.25×10^{5}	50 percent decrease in O_2 evolution
Dichlobenil	Technical acid	Dunaliella tertiolecta			6×10^{4}	50 percent decrease in growth
Dichlobenil	Technical acid	Isochrysis galbana			1×10^{5}	50 percent decrease in O_2 evolution
Dichlobenil	Technical acid	Isochrysis galbana			6×10^{4}	50 percent decrease in growth
Dichlobenil	Technical acid	Phaeodactylum tricornutum			1.5×10^{5}	50 percent decrease in O_2 evolution
Dichlobenil	Technical acid	Phaeodactylum tricornutum			2.5×10^{4}	50 percent decrease in growth
Organochlorine						
MCPA		Crassostrea virginica	American oyster	Egg	1.962×10^{4}	TLM
MCPA		Crassostrea virginica	American oyster	Larvae	3.13×10^{4}	TLM
Phenoxyacetic acid						
2,4-D	Ester	Crassostrea virginica	American oyster	Egg	8×10^{3}	TLM
2,4-D	Ester	Crassostrea virginica	American oyster	Larvae	740	TLM
2,4-D	Salt	Crassostrea virginica	American oyster	Egg	2.044×10^{4}	TLM
2,4-D	Salt	Crassostrea virginica	American oyster	Larvae	6.429×10^{4}	TLM
2,4-D	Technical acid	Chlorococcum sp.			6×10^{4}	50 percent decrease in O_2 evolution
2,4-D	Technical acid	Chlorococcum sp.			5×10^{4}	50 percent decrease in growth
2,4-D	Technical acid	Dunaliella tertiolecta			9×10^{4}	50 percent decrease in O_2 evolution
2,4-D	Technical acid	Dunaliella tertiolecta			7.5×10^{4}	50 percent decrease in growth
2,4-D	Technical acid	Isochrysis galbana			6×10^{4}	50 percent decrease in O_2 evolution
2,4-D	Technical acid	Isochrysis galbana			5×10^{4}	50 percent decrease in growth

Continued

Test Procedure	Temp C	Salinity ‰	Other Environmental Criteria	Statistical Evaluation	Residue levels mg/kg	Other Parameters	Reference
f	20	30	pH = 7.9-8.1 6000 lux 12/12	"			Walsh "
Growth measured as ABS. (525 mu) after 10 days	20	30	pH = 7.9-8.1 6000 lux 12/12	"			Walsh "
f	20	30	pH = 7.9-8.1 6000 lux 12/12	"			Walsh "
Growth measured as ABS. (525 mu) after 10 days	20	30	pH = 7.9-8.1 6000 lux 12/12	"			Walsh "
f	20	30	pH = 7.9-8.1 6000 lux 12/12	"			Walsh "
Growth measured as ABS. (525 mu) after 10 days	20	30	pH = 7.9-8.1 6000 lux 12/12	"			Walsh "
f	20	30	pH = 7.9-8.1 6000 lux 12/12	"			Walsh "
Growth measured as ABS. (525 mu) after 10 days	20	30	pH = 7.9-8.1 6000 lux 12/12	"			Walsh "
f	20	30	pH = 7.9-8.1 6000 lux 12/12	"			Walsh 1972[949]
Growth measured as ABS. (525 mu) after 10 days	20	30	pH = 7.9-8.1 6000 lux 12/12	"			Walsh "
f	20	30	pH = 7.9-8.1 6000 lux 12/12	"			Walsh "
Growth measured as ABS. (525 mu) after 10 days	20	30	pH = 7.9-8.1 6000 lux 12/12	"			Walsh "
f	20	30	pH = 7.9-8.1 6000 lux 12/12	"			Walsh "
Growth measured as ABS. (525 mu) after 10 days	20	30	pH = 7.9-8.1 6000 lux 12/12	"			Walsh "
f	20	30	pH 7.9-8.1 6000 lux 12/12	Litchfield & Wilcoxon Method[277]			Walsh 1972[949]
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9-8.1 6000 lux 12/12	"			Walsh "
f	20	30	pH 7.9-8.1 6000 lux 12/12	"			Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9-8.1 6000 lux 12/12	"			Walsh "
f	20	30	pH 7.9-8.1 6000 lux 12/12	"			Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9-8.1 6000 lux 12/12	"			Walsh "
f	20	30	pH 7.9-8.1 6000 lux 12/12	"			Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9-8.1 6000 lux 12/12	"			Walsh "
f	20	30	pH 7.9-8.1 6000 lux 12/12	"			Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9-8.1 6000 lux 12/12	"			Walsh "
f	20	30	pH 7.9-8.1 6000 lux 12/12	"			Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9-8.1 6000 lux 12/12	"			Walsh "
f	20	30	pH 7.9-8.1 6000 lux 12/12	"			Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9-8.1 6000 lux 12/12	"			Walsh "
f	20	30	pH 7.9-8.1 6000 lux 12/12	"			Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9-8.1 6000 lux 12/12	"			Walsh "
f	20	30	pH 7.9-8.1 6000 lux 12/12	"			Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9-8.1 6000 lux 12/12	"			Walsh "
f	20	30	pH 7.9-8.1 6000 lux 12/12	"			Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9-8.1 6000 lux 12/12	"			Walsh "
48 hr static lab bioassay	24±1			None			Davis and Hidu 1969[234]
14 day static lab bioassay	24±1			None			Davis and Hidu 1969[234]
48 hr static lab bioassay	24±1			None			Davis and Hidu 1965[234]
14 day static lab bioassay	24±1			None			Davis and Hidu 1969[234]
48 hr static lab bioassay	24±1			None			Davis and Hidu 1965[234]
14 day static lab bioassay	24±1			None			Davis and Hidu 1965[234]
f	20	30	pH 7.9-8.1 6000 lux 12/12	Litchfield & Wilcoxon Method[277]			Walsh 1972[949]
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9-8.1 6000 lux 12/12	"			Walsh "
f	20	30	pH 7.9-8.1 6000 lux 12/12	"			Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9-8.1 6000 lux 12/12	"			Walsh "
f	20	30	pH 7.9-8.1 6000 lux 12/12	"			Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9-8.1 6000 lux 12/12	"			Walsh "

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f O_2 evolution measured by Gilson differental respirometer on 4 ml of culture in log phase. Length of test 90 minutes.

TABLE 6—

Substance Tested	Formulation	Organism Tested	Common Name	Life Stage or Size (mm)	Conc. (ppb act ingred.) in water	Methods of Assessment
2,4-D	Technical acid	Phaeodactylum tricornutum			6×10^4	50 percent decrease in O_2 evolution
2,4-D	Technical acid	Phaeodactylum tricornutum			5×10^4	50 percent decrease in growth
2,4-D	Butoxyethanol ester	Chlorococcum sp.			1×10^5	50 percent decrease in O_2 evolution
2,4-D	Butoxyethanol ester	Chlorococcum sp.			7.5×10^4	50 percent decrease in growth
2,4-D	Butoxyethanol ester	Dunaliella tertiolecta			1×10^5	50 percent decrease in O_2 evolution
2,4-D	Butoxyethanol ester	Dunaliella tertiolecta			7.5×10^4	50 percent decrease in growth
2,4-D	Butoxyethanol ester	Isochrysis galbana			1×10^5	50 percent decrease in O_2 evolution
2,4-D	Butoxyethanol ester	Isochrysis galbana			7.5×10^4	50 percent decrease in growth
2,4-D	Butoxyethanol ester	Phaeodactylum tricornutum			2×10^5	50 percent decrease in O_2 evolution
2,4-D	Butoxyethanol ester	Phaeodactylum tricornutum			1.5×10^5	50 percent decrease in growth
EMID	2,4-D cmpd	Crassostrea virginica	American oyster	Eggs	1.682×10^4	TLM
EMID	2,4-D cmpd	Crassostrea virginica	American oyster	Larvae	3.0×10^4	TLM
2,4,5-T	Technical acid	Chlorococcum sp.			1.5×10^5	50 percent decrease in O_2 evolution
2,4,5-T	Technical acid	Chlorococcum sp.			1.0×10^5	50 percent decrease in growth
2,4,5-T	Technical acid	Dunaliella tertiolecta			1.5×10^5	50 percent decrease in O_2 evolution
2,4,5-T	Technical acid	Dunaliella tertiolecta			1.25×10^5	50 percent decrease in growth
2,4,5-T	Technical acid	Isochrysis galbana			5×10^4	50 percent decrease in O_2 evolution
2,4,5-T	Technical aicd	Isochrysis galbana			5×10^4	50 percent decrease in growth
2,4,5-T	Technical acid	Phaeodactylum tricornutum			7.5×10^4	50 percent decrease in O_2 evolution
2,4,5-T	Technical acid	Phaeodactylum tricornutum			5×10^4	50 percent decrease in growth
Phthalic						
Endothall	Technical acid	Chlorococcum sp.			1×10^5	50 percent decrease in O_2 evolution
Endothall	Technical acid	Chlorococcum sp.			5×10^4	50 percent decrease in growth
Endothall	Technical acid	Dunaliella tertiolecta			4.25×10^5	50 percent decrease in O_2 evolution
Endothall	Technical acid	Dunaliella tertiolecta			5×10^4	50 percent decrease in growth
Endothall	Technical acid	Isochrysis galbana			6×10^4	50 percent decrease in O_2 evolution
Endothall	Technical acid	Isochrysis galbana			2.5×10^4	50 percent decrease in growth
Endothall	Technical acid	Phaeodactylum tricornutum			7.5×10^4	50 percent decrease in O_2 evolution
Endothall	Technical acid	Phaeodactylum tricornutum			1.5×10^4	50 percent decrease in growth
Endothall	Amine salt	Chlorococcum sp.			$>1\times10^6$	50 percent decrease in O_2 evolution
Endothall	Amine salt	Chlorococcum sp.			3×10^5	50 percent decrease in growth
Endothall	Amine salt	Dunaliella tertiolecta			$>1\times10^5$	50 percent decrease in O_2 evolution
Endothall	Amine salt	Dunaliella tertiolecta			4.5×10^4	50 percent decrease in growth
Endothall	Amine salt	Isochrysis galbana			$>1\times10^6$	50 percent decrease in O_2 evolution
Endothall	Amine salt	Isochrysis galbana			2.25×10^4	50 percent decrease in growth
Endothall	Amine salt	Phaeodactylum tricornutum			$>1\times10^6$	50 percent decrease in O_2 evolution
Endothall	Amine salt	Phaeodactylum tricornutum			2.5×10^4	50 percent decrease in growth
Endothall		Crassostrea virginica	American oyster	Egg	2.822×10^4	TLM
Endothall		Crassostrea virginica	American oyster	Larvae	4.800×10^4	TLM
Endothall		Mercenaria mercenaria	Hard clam	Egg	5.102×10^4	TLM
Endothall		Mercenaria mercenaria	Hard clam	Larvae	1.25×10^4	TLM
Picolinic acid						
Tordon® 101		Chlorococcum sp.			$>2\times10^6$	50 percent decrease in O_2 evolution
Tordon® 101		Chlorococcum sp.			1×10^5	50 percent decrease in growth
Tordon® 101		Dunaliella tertiolecta			$>2\times10^6$	50 percent decrease in O_2 evolution
Tordon® 101		Dunaliella tertiolecta			1.25×10^5	50 percent decrease in growth
Tordon® 101		Isochrysis galbana			1×10^5	50 percent decrease in O_2 evolution
Tordon® 101		Isochrysis galbana			5×10^4	50 percent decrease in growth
Tordon® 101		Phaeodactylum tricornutum			$>72\times10^4$	50 percent decrease in O_2 evolution
Tordon® 101		Phaeodactylum tricornutum			1×10^5	50 percent decrease in growth

Continued

Test Procedure	Temp C	Salinity °/oo	Other Environmental Criteria	Statistical Evaluation	Residue levels mg/kg	Other Parameters	Reference
/	20	30	pH 7.9–8.1 6000 lux 12/12	"	....	....	Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"	....	....	Walsh "
/	20	30	pH 7.9–8.1 6000 lux 12/12	"	....	....	Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"	....	....	Walsh "
/	20	30	pH 7.9–8.1 6000 lux 12/12	"	....	....	Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"	....	....	Walsh "
/	20	30	pH 7.9–8.1 6000 lux 12/12	"	....	....	Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"	....	....	Walsh "
/	20	30	pH 7.9–8.1 6000 lux 12/12	"	....	....	Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"	....	....	Walsh "
48 hr static lab bioassay	24±1	....	....	....	....	....	Davis and Hidu 1969[724]
14 day static lab bioassay	24±1	....	....	....	....	....	Davis and Hidu 1969[724]
/	20	30	pH 7.9–8.1 6000 lux 12/12	Litchfield & Wilcoxon Method[723]	....	....	Walsh 1972[949]
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"	....	....	Walsh "
/	20	30	pH 7.9–8.1 6000 lux 12/12	"	....	....	Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"	....	....	Walsh "
/	20	30	pH 7.9–8.1 6000 lux 12/12	"	....	....	Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"	....	....	Walsh "
/	20	30	pH 7.9–8.1 6000 lux 12/12	"	....	....	Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"	....	....	Walsh "
/	20	30	pH 7.9–8.1 6000 lux 12/12	Litchfield & Wilcoxon Method[723]	....	....	Walsh 1972[949]
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"	....	....	Walsh "
/	20	30	pH 7.9–8.1 6000 lux 12/12	"	....	....	Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"	....	....	Walsh "
/	20	30	pH 7.9–8.1 6000 lux 12/12	"	....	....	Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"	....	....	Walsh "
/	20	30	pH 7.9–8.1 6000 lux 12/12	"	....	....	Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"	....	....	Walsh "
/	20	30	pH 7.9–8.1 6000 lux 12/12	"	....	....	Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"	....	....	Walsh "
/	20	30	pH 7.9–8.1 6000 lux 12/12	"	....	....	Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"	....	....	Walsh "
/	20	30	pH 7.9–8.1 6000 lux 12/12	"	....	....	Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"	....	....	Walsh "
/	20	30	pH 7.9–8.1 6000 lux 12/12	"	....	....	Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"	....	....	Walsh "
48 hr static lab bioassay	24±1	....	....	None	....	....	Davis and Hidu 1969[724]
14 day static lab bioassay	24±1	....	....	None	....	....	Davis and Hidu 1969[724]
48 hr static lab bioassay	24±1	....	....	None	....	....	Davis and Hidu 1969[724]
12 day static lab bioassay	24±1	....	....	None	....	....	Davis and Hidu 1969[724]
/	20	30	pH 7.9–8.1 6000 lux 12/12	Litchfield & Wilcoxon Method[723]	....	....	Walsh 1972[949]
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"	....	....	Walsh "
/	20	30	pH 7.9–8.1 6000 lux 12/12	"	....	....	Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"	....	....	Walsh "
/	20	30	pH 7.9–8.1 6000 lux 12/12	"	....	....	Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"	....	....	Walsh "
/	20	30	pH 7.9–8.1 6000 lux 12/12	"	....	....	Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"	....	....	Walsh "

O_2 evolution measured by Gilson differential respirometer on 4 ml of culture in log phase. Test length 90 minutes

TABLE 6—

Substance Tested	Formulation	Organism Tested	Common Name	Life Stage or Size (mm)	Conc. (ppb act. ingred.) in water	Methods of Assessment
Propionic acid						
Dalapon	Technical acid	Chlorococcum sp.			2.5×10^4	50 percent decrease in O_2 evolution
Dalapon	Technical acid	Chlorococcum sp.			5×10^4	50 percent decrease in growth
Dalapon	Technical acid	Dunaliella tertiolecta			2.5×10^4	50 percent decrease in O_2 evolution
Dalapon	Technical acid	Dunaliella tertiolecta			$1.\times10^4$	50 percent decrease in growth
Dalapon	Technical acid	Isochrysis galbana			4×10^4	50 percent decrease in O_2 evolution
Dalapon	Technical acid	Isochrysis galbana			2×10^4	50 percent decrease in growth
Dalapon	Technical acid	Phaeodactylum tricornutum			2.5×10^4	50 percent decrease in O_2 evolution
Dalapon	Technical acid	Phaeodactylum tricornutum			2.5×10^4	50 percent decrease in growth
Silvex	Technical acid	Chlorococcum sp.			2.5×10^4	50 percent decrease in O_2 evolution
Silvex	Technical acid	Chlorococcum sp.			2.5×10^4	50 percent decrease in growth
Silvex	Technical acid	Dunaliella tertiolecta			2×10^4	50 percent decrease in O_2 evolution
Silvex	Technical acid	Dunaliella tertiolecta			2.5×10^4	50 percent decrease in growth
Silvex	Technical acid	Isochrysis galbana			2.5×10^4	50 percent decrease in O_2 evolution
Silvex	Technical acid	Isochrysis galbana			5×10^3	50 percent decrease in growth
Silvex		Crassostrea virginica	American oyster	Egg	5.9×10^3	TLM
Silvex		Crassostrea virginica	American oyster	Larvae	710	TLM
Toluidine						
Trifluralin	Technical acid	Chlorococcum sp.			5×10^4	50 percent decrease in O_2 evolution
Trifluralin	Technical acid	Chlorococcum sp.			2.5×10^3	50 percent decrease in growth
Trifluralin	Technical acid	Dunaliella tertiolecta			$>5\times10^4$	50 percent decrease in O_2 evolution
Trifluralin	Technical acid	Dunaliella tertiolecta			5×10^3	50 percent decrease in growth
Trifluralin	Technical acid	Isochrysis galbana			4×10^4	50 percent decrease in O_2 evolution
Trifluralin	Technical acid	Isochrysis galbana			2.5×10^3	50 percent decrease in growth
Trifluralin	Technical acid	Phaeodactylum tricornutum			$>5\times10^4$	50 percent decrease in O_2 evolution
Trifluralin	Technical acid	Phaeodactylum tricornutum			2.5×10^3	50 percent decrease in growth
Triazine						
Ametryne	Technical acid	Chlorococcum sp.			20	50 percent decrease in O_2 evolution
Ametryne	Technical acid	Chlorococcum sp.			10	50 percent decrease in growth
Ametryne	Technical acid	Dunaliella tertiolecta			40	50 percent decrease in O_2 evolution
Ametryne	Technical acid	Dunaliella tertiolecta			40	50 percent decrease in growth
Ametryne	Technical acid	Isochrysis galbana			10	50 percent decrease in O_2 evolution
Ametryne	Technical acid	Isochrysis galbana			10	50 percent decrease in growth
Ametryne	Technical acid	Phaeodactylum tricornutum			10	50 percent decrease in O_2 evolution
Ametryne	Technical acid	Phaeodactylum tricornutum			20	50 percent decrease in growth
Atrazine	Technical acid	Chlorococcum sp.			100	50 percent decrease in O_2 evolution
Atrazine	Technical acid	Chlorococcum sp.			100	50 percent decrease in growth
Atrazine	Technical acid	Dunaliella tertiolecta			300	50 percent decrease in O_2 evolution
Atrazine	Technical acid	Dunaliella tertiolecta			300	50 percent decrease in growth
Atrazine	Technical acid	Isochrysis galbana			100	50 percent decrease in O_2 evolution
Atrazine	Technical acid	Isochrysis galbana			100	50 percent decrease in growth
Atrazine	Technical acid	Phaeodactylum tricornutum			100	50 percent decrease in O_2 evolution
Atrazine	Technical acid	Phaeodactylum tricornutum			200	50 percent decrease in growth
Prometone	Technical acid	Chlorococcum sp.			400	50 percent decrease in O_2 evolution
Prometone	Technical acid	Chlorococcum sp.			500	50 percent decrease in growth
Prometone	Technical acid	Dunaliella tertiolecta			2×10^3	50 percent decrease in O_2 evolution
Prometone	Technical acid	Dunaliella tertiolecta			1.5×10^3	50 percent decrease in growth
Prometone	Technical acid	Isochrysis galbana			1×10^3	50 percent decrease in O_2 evolution
Prometone	Technical acid	Isochrysis galbana			1×10^3	50 percent decrease in growth

Continued

Test Procedure	Temp C	Salinity ‰	Other Environmental Criteria	Statistical Evaluation	Residue levels mg/kg	Other Parameters	Reference
f	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
f	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
f	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
f	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
f	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
f	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
f	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
48 hr static lab bioassay	24±1			None			Davis and Hidu 1969[224]
14 day static lab bioassay	24±1			None			Davis and Hidu 1969[224]
f	20	30	pH 7.9–8.1 6000 lux 12/12	Litchfield & Wilcoxon Method[837]			Walsh 1972[948]
Measured as ABS (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
f	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
Measured as ABS (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
f	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
Measured as ABS (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
f	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
Measured as ABS (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
f	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
Measured as ABS (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
f	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
Measured as ABS (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
f	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
Measured as ABS (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
f	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
Measured as ABS (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
f	20	30	pH 7.9–8.1 6000 lux 12/12	Litchfield & Wilcoxon Method[837]			Walsh 1972[948]
Measured as ABS (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12				Walsh "
f	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
Measured as ABS (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
f	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
Measured as ABS (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
f	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
Measured as ABS (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
f	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
f	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
f	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9–8.1 6000 lux 12/12	"			Walsh "

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f O_2 evolution measured by Gilson Differential Respirometer on 4 ml of culture in log phase. Length of test 90 min.

TABLE 6—

Substance Tested	Formulation	Organism Tested	Common Name	Life Stage or Size (mm)	Conc. (ppb act. ingred.) in water	Methods of Assessment
Prometone	Technical acid	Phaeodactylum tricornutum			100	50 percent decrease in O_2 evolution
Prometone	Technical acid	Phaeodactylum tricornutum			250	50 percent decrease in growth
Simazine	Technical acid	Chlorococcum sp.			2.5×10^3	50 percent decrease in O_2 evolution
Simazine	Technical acid	Chlorococcum sp.			2×10^3	50 percent decrease in growth
Simazine	Technical acid	Dunaliella tertiolecta			4×10^3	50 percent decrease in O_2 evolution
Simazine	Technical acid	Dunaliella tertiolecta			5×10^3	50 percent decrease in growth
Simazine	Technical acid	Isochrysis galbana			600	50 percent decrease in O_2 evolution
Simazine	Technical acid	Isochrysis galbana			500	50 percent decrease in growth
Simazine	Technical acid	Phaeodactylum tricornutum			600	50 percent decrease in O_2 evolution
Simazine	Technical acid	Phaeodactylum tricornutum			500	50 percent decrease in growth
Herbicides Substituted urea compounds						
Diuron		Protococcus			0.02	.52 OPT. DEN. expt/OPT DEN control
Diuron		Chlorella sp			4.00	.34 OPT DEN. expt/OPT DEN control
Diuron	Technical	Dicrateria inornata			[g]	32.3 percent decrease $(CH_2O)x$
Diuron	Technical	Nanochloris sp			[g]	18.8 decrease $(CH_2O)x$
Diuron	Technical	Chlorococcum sp			10	61 percent inhibition of growth
Diuron	Technical	Chlorococcum sp			[g]	65.6 inhibition (CH_2O)
Diuron	Technical acid	Chlorococcum sp			20	50 percent reduction O_2 evolution[f]
Diuron	Technical acid	Chlorococcum sp			10	50 percent reduction in growth
Diuron	Technical acid	Dunaliella tertiolecta			10	50 percent reduction O_2 evolution[f]
Diuron	Technical acid	Dunaliella tertiolecta			20	50 percent reduction in growth
Diuron	Technical	Dunaliella tertiolecta			[g]	17.9 percent decrease $(CH_2O)x$
Diuron		Dunaliella euchlora			0.4	.44 OPT. DEN. expt/OPT. DEN control
Diuron	Technical	Isochrysis galbana			[g]	37.4 percent decrease $(CH_2O)x$
Diuron	Technical acid	Isochrysis galbana			10	50 percent reduction O_2 evolution[f]
Diuron	Technical acid	Isochrysis galbana			10	50 percent reduction in growth
Diuron	Technical	Monochrysis lutheri			[g]	35.7 percent decrease (CH_2O)
Diuron		Monochrysis lutheri			0.02	.00 optical density expt/optical density control
Diuron		Phaeodactylum tricornutum			0.4	.79 OPT. DEN expt/OPT DEN control
Diuron		Phaeodactylum tricornutum			4.0	.00 OPT. DEN expt/OPT DEN control
Diuron	Technical acid	Phaeodactylum tricornutum			10.	50 percent reduction O_2 evolution[f]
Diuron	Technical acid	Phaeodactylum tricornutum			10.	50 percent reduction in growth
Fenuron		Protococcus sp.			2,900	.33 Opt. Den. Expt/Opt. Den Control
Fenuron		Chlorella sp			290	.82 Opt. Den. Expt/Opt. Den. Control
Fenuron		Chlorella sp			2,900	.00 Opt. Den. Expt/Opt. Den. Control[h]
Fenuron	Technical acid	Chlorococcum sp.			1,000	68 percent inhibition of growth
Fenuron	Technical acid	Chlorococcum sp.			750	50 percent decrease in growth
Fenuron	Technical acid	Chlorococcum sp.			2,000	50 percent decrease in O_2 evolution
Fenuron	Technical acid	Dunaliella tertiolecta			1,250	50 percent decrease in O_2 evolution
Fenuron	Technical acid	Dunaliella tertiolecta			1,500	50 percent decrease in growth
Fenuron		Dunaliella euchlora			290	.46 Opt. Den. Expt/Opt. Den. Control
Fenuron	Technical acid	Isochrysis galbana			1,250	50 percent decrease O_2 evolution
Fenuron	Technical acid	Isochrysis galbana			750	50 percent decrease growth
Fenuron		Monochrysis lutheri			290	.67 Opt. Den. Expt/Opt. Den. Control
Fenuron		Monochrysis lutheri			2,900	.00 Opt. Den Expt/Opt. Den. Control
Fenuron		Phaeodactylum tricornutum			290	.82 Opt. Den. Expt/Opt Den. Control
Fenuron	Technical acid	Phaeodactylum tricornutum			1,250	50 percent decrease O_2 evolution
Fenuron	Technical acid	Phaeodactylum tricornutum			750	50 percent decrease growth
Monuron		Protococcus sp.			1.	.90 OD expt/OD control
Monuron		Protococcus sp.			20	.00 OD expt/OD control[h]
Monuron		Chlorella sp.			1.	.30 OD expt/OD control
Monuron	Technical acid	Chlorococcum sp.			100	54 percent inhibition of growth
Monuron	Technical acid	Chlorococcum sp.			100	50 percent decrease O_2 evolution
Monuron	Technical aicd	Chlorococcum sp.			100	50 percent decrease in growth
Monuron	Technical acid	Dunaliella tertiolecta			90	50 percent decrease O_2 evolution
Monuron	Technical acid	Dunaliella tertiolecta			150	50 percent decrease growth
Monuron		Dunaliella euchlora			1	1.00 OD expt/OD control
Monuron		Dunaliella euchlora			20	.00 OD expt/OD control[h]
Monuron	Technical acid	Isochrysis galbana			100	50 percent decrease O_2 evolution
Monuron	Technical acid	Isochrysis galbana			130	50 percent decrease in growth
Monuron		Monochrysis lutheri			1	.83 OD expt/OD control
Monuron	Technical acid	Phaeodactylum tricornutum			90	50 percent decrease O_2 evolution
Monuron	Technical a-id	Phaeodactylum tricornutum			100	50 percent decrease in growth
Monuron		Phaeodactylum tricornutum			1	.65 OD expt/OD control
Monuron		Phaeodactylum tricornutum			20	.00 OD expt/OD control

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[f] O_2 evolution measured with a Gilson differential despirometer on 4 ml of culture in log-phase.

[g] Conc. which decrease growth by 50–75 percent as determined by Walsh and Grow Diuron 10 ppb; fenuron 1000 ppb; monuron 100 ppb; neburon 30 ppb.

[h] No growth but organisms viable.

Continued

Test Procedure	Temp C	Salinity ‰	Other Environmental Criteria	Statistical Evaluation	Residue levels mg/kg	Other Parameters	Reference
f	20	30	pH 7.9-8.1 6000 lux 12/12	"	...	...	Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9-8.1 6000 lux 12/12	"	...	...	Walsh "
f	20	30	pH 7.9-8.1 6000 lux 12/12	"	...	...	Walsh "
Measured as ABS. (525 mu) after 10 days	20	30	pH 7.9-8.1 6000 lux 12/12	"	...	...	Walsh "
f	20	30	pH 7.9-8.1 6000 lux 12/12	"	...	...	Walsh "
Measured as APS. (525 mu) after 10 days	20	30	pH 7.9-8.1 6000 lux 12/12	"	...	...	Walsh "
f	20	30	pH 7.9-8.1 6000 lux 12/12	"	...	...	Walsh "
Measured as APS. (525 mu) after 10 days	20	30	pH 7.9-8.1 6000 lux 12/12	"	...	...	Walsh "
f	20	30	pH 7.9-8.1 6000 lux 12/12	"	...	...	Walsh "
Measured as APS. (525 mu) after 10 days	20	30	pH 7.9-8.1 6000 lux 12/12	"	...	...	Walsh "
10 day growth test	20±.5	...	...	None	...	...	Ukeles 1962^{547}
10 day growth test	20±.5	...	...	None	...	...	Ukeles 1962^{547}
10 dav growth test	20	30	pH 7.9-8.1 6000 lux 12/12	Significant at 0.05 level	...	...	Walsh and Grow 1971^{549}
10 day growth test	20	30	pH 7.9-8.1 6000 lux 12/12	Significant at 0.05 level	...	...	Walsh and Grow 1971^{549}
10 day growth test	20	30	pH 7.9-8.1 6000 lux 12/12	None	...	...	Walsh and Grow 1971^{549}
10 day growth test	20	30	pH 7.9-8.1 6000 lux 12/12	Significant at 0.05 level	...	...	Walsh and Grow 1971^{549}
10 day growth test	20	30	pH 7.9-8.1 6000 lux 12/12	Litchfield & Wilcoxon method^{337}	...	...	Walsh 1972^{548}
10 day growth test	20	30	pH 7.9-8.1 6000 lux 12/12	"	...	...	Walsh "
10 day growth test	20	30	pH 7.9-8.1 6000 lux 12/12	"	...	...	Walsh "
10 day growth test	20	30	pH 7.9-8.1 6000 lux 12/12	Significant at 0.05 level	...	...	Walsh "
10 day growth test	20	30	pH 7.9-8.1 6000 lux 12/12	None	...	...	Walsh and Grow 1971^{549}
10 day growth test	20±.5	...	...	None	...	...	Ukeles ^{547}
10 day growth test	20	30	pH 7.9-8.1 6000 lux 12/12	Significant at 0.05 level	...	...	Walsh and Grow 1971^{549}
10 day growth test	20	30	pH 7.9-8.1 6000 lux 12/12	Litchfield & Wilcoxon method^{337}	...	...	Walsh 1972^{548}
10 day growth test	20	30	pH 7.9-8.1 6000 lux 12/12	"	...	...	Walsh "
10 day growth test	20	30	pH 7.9-8.1 6000 lux 12/12	Significant at 0.05 level	...	...	Walsh and Grow 1971^{549}
10 day growth test	20±.5	...	...	...	...	...	Ukeles 1962^{547}
10 day growth test	20±.5	...	...	None	...	...	Ukeles 1962^{547}
10 day growth test	20±.5	...	...	None	...	...	Ukeles 1962^{547}
10 day growth test	20	30	pH 7.9-8.1 6000 lux 12/12	Litchfield and Wilcoxon method^{337}	...	...	Walsh 1972^{548}
10 day growth test	20	30	pH 7.9-8.1 6000 lux 12/12	"	...	...	Walsh "
10 day growth test	20.5±1	...	500 ft.-c continuous	None	...	...	Ukeles 1962^{547}
10 day growth test	20.5±1	...	500 ft.-c continuous	None	...	...	Ukeles 1962^{547}
10 day growth test	20.5±1	...	500 ft.-c continuous	None	...	...	Ukeles 1962^{547}
10 day growth test	20	30	pH 7.9-8.1 6000 lux 12/12	None	...	...	Walsh and Grow 1971^{549}
10 day growth test	20	30	pH 7.9-8.1 6000 lux 12/12	Litchfield & Wilcoxon method^{337}	...	...	Walsh 1972^{548}
f	20	30	pH 7.9-8.1	"	...	...	Walsh "
f	20	30	pH 7.9-8.1	"	...	...	Walsh "
10 day growth test	20	30	pH 7.9-8.1 6000 lux 12/12	"	...	...	Walsh 1972^{548}
10 day growth test	20.5±1	...	500 ft.-c continuous	"	...	...	Ukeles 1962^{547}
f	20	30	pH 7.9-8.1	"	...	...	Walsh 1972^{548}
10 day growth test	20	30	pH 7.9-8.1 6000 lux 12/12	None	...	...	Walsh "
10 day growth test	20.5±1	...	500 ft.-c continuous	None	...	...	Ukeles 1962^{547}
10 day growth test	20.5±1	...	500 ft.-c continuous	None	...	...	Ukeles 1962^{547}
10 day growth test	20.5±1	30	500 ft.-c continuous	None	...	...	Ukeles 1962^{547}
f	20	30	pH 7.9-8.1 6000 lux 12/12	Litchfield & Wilcoxon method^{337}	...	...	Walsh 1972^{548}
10 day growth test	20	30	pH 7.9-8.1 6000 lux 12/12	"	...	...	Walsh "
10 day growth test	20.5±1	...	500 ft.-c continuous	None	...	...	Ukeles 1962^{547}
10 day growth test	20.5±1	...	500 ft.-c continuous	None	...	...	Ukeles 1962^{547}
10 day growth test	20.5±1	...	500 ft.-c continuous	None	...	...	Ukeles 1962^{547}
10 day growth test	20	30	pH 7.9-8.1 6000 lux 12/12	None	...	...	Walsh and Grow 1971^{549}
f	20	30	pH=7.9-8.1	Litchfield & Wilcoxon Method^{337}	...	...	Walsh 1972^{548}
10 day growth test	.0	30	pH=7.9-8.1 6000 lux 12/12	"	...	...	Walsh "
f	20	30	pH=7.9-8.1	"	...	...	Walsh "
10 day growth test	20	30	pH=7.9-8.1 6000 lux 12/12	"	...	...	Walsh "
10 day growth test	20.5±1	...	500 ft.-c continuous	None	...	...	Ukeles 1962^{547}
10 day growth test	20.5±1	...	500 ft.-c continuous	Litchfield & Wilcoxon Method^{337}	...	...	Walsh 1972^{548}
f	20	30	pH=7.9-8.1	"	...	...	Walsh "
10 day growth test	20	30	pH=7.9-8.1 6000 lux 12/12	"	...	...	Walsh "
10 day growth test	20.5±1	...	500 ft.-c continuous	None	...	...	Ukeles 1962^{547}
f	20	30	pH=7.9-8.1	Litchfield & Wilcoxon Method^{337}	...	...	Walsh 1972^{548}
10 day growth test	20	30	pH=7.9-8.1 6000 lux 12/12	"	...	...	Walsh "
10 day growth test	20.5±1	...	500 ft.-c continuous	None	...	...	Ukeles 1962^{547}
10 day growth test	20.5±1	...	500 ft.-c continuous	None	...	...	Ukeles 1962^{547}

TABLE 6—

Substance Tested	Formulation	Organism Tested	Common Name	Life Stage or Size (mm)	Conc. (ppb act. ingred.) in water	Methods of Assessment
Neburon		Protococcus sp.			40	.41 OD expt/OD control
Neburon		Chlorella sp.			40	.31 OD expt/OD control
Neburon	Technical acid	Chlorococcum sp.			30	68 percent inhibition in growth
Neburon	Technical acid	Chlorococcum sp.			20	50 percent decrease O_2 evolution
Neburon	Technical acid	Chlorococcum sp.			30	50 percent decrease growth
Neburon	Technical acid	Dunaliella tertiolecta			20	50 percent decrease O_2 evolution
Neburon	Technical acid	Dunaliella tertiolecta			40	50 percent decrease growth
Neburon		Dunaliella euchlora			40	.47 OD expt/OD control
Neburon	Technical acid	Isochrysis galbana			20	50 percent decrease O_2 evolution
Neburon	Technical acid	Isochrysis galbana			30	50 percent decrease growth
Neburon		Monochrysis lutheri			40	.00 OD expt/OD control
Neburon		Phaeodactylum tricornutum			40	.10 OD expt/OD control
Neburon	Technical acid	Phaeodactylum tricornutum			40	50 percent decrease O_2 evolution
Neburon	Technical acid	Phaeodactylum tricornutum			30	50 percent decrease growth
Bactericides, Fungicides Nematocides, and misc.						
Aroclor	1254	Tetrahymena pyriformis		Log-phase	10	13.30 percent decrease in population size measure at 540 $m\mu$
Aroclor	1254	Penaeus duorarum	Pink shrimp	25–38	0.94	51 percent mortality
Aroclor	1254	Penaeus duorarum	Pink shrimp	95–125	3.5	50 percent mortality
Aroclor	1254	Leiostomus xanthurus	Spot	24	5	50 percent mortality
Aroclor	1254	Lagodon rhomboides	Pinfish	30	5	50 percent mortality
Chloramphenicol		Mercenaria mercenaria	Hard clam	Egg	7.429×10^4	TLM
Chloramphenicol		Mercenaria mercenaria	Hard clam	Larvae	$5.\times10^4$	TLM
Delrad		Mercenaria mercenaria	Hard clam	Larvae	72	TLM
Delrad		Crassostrea virginica	American oyster	Larvae	31	TLM
Dowacide A	97 percent	Protococcus sp			2.5×10^4	.75 O D expt/O D control
Dowacide A	97 percent	Chlorella sp.			5×10^4	.74 O.D. expt/O.D. control
Dowacide A	97 percent	Dunaliella euchlora			5×10^3	.52 O.D. expt/O.D. control
Dowacide A	97 percent	Phaeodactylum tricornutum			2.5×10^4	.48 O.D. expt/O.D. control
Dowacide A	97 percent	Monochrysis lutheri			2.5×10^4	.22 O.D. expt/O.D. control
Dowacide A	97 percent	Mercenaria mercenaria	Hard clam	Eggs	1×10^5	TLM
Dowacide A	97 percent	Mercenaria mercenaria	Hard clam	Larvae	750	TLM
Dowacide G		Mercenaria mercenaria	Hard clam	Eggs	<250	TLM
Dowacide G		Mercenaria mercenaria	Hard clam	Larvae	<250	TLM
Giseofulvin		Mercenaria mercenaria	Hard clam	Egg	<250	TLM
Giseofulvin		Mercenaria mercenaria	Hard clam	Larvae	$<1.\times10^3$	TLM
Lignasan	6.25 percent	Protococcus sp.			6	.00 O.D. expt/O.D. control
Lignasan	6.25 percent	Chlorella sp.			6	.00 O.D. expt/O.D. control
Lignasan	6.25 percent	Dunaliella euchlora			6	.31 O.D. expt/O.D. control
Lignasan	6.25 percent	Phaeodactylum tricornutum			0.6	.55 O.D. expt/O.D. control
Lignasan	6.25 percent	Monochrysis lutheri			6	.00 O.D. expt/O.D. control
Nabam		Protococcus sp.			1×10^3	.53 O.D. expt/O.D. control
Nabam		Chlorella sp.			1×10^3	.63 O.D. expt/O.D. control
Nabam		Dunaliella euchlora			100	.27 O.D. expt/O.D. control
Nabam		Phaeodactylum tricornutum			1×10^3	.00 O.D. expt/O.D. control[a]
Nabam		Monochrysis lutheri			100	.48 O.D. expt/O.D. control
Nabam		Mercenaria mercenaria	Hard clam	Egg	<500	TLM
Nabam		Mercenaria mercenaria	Hard clam	Larvae	1.75×10^3	TLM
Nabam		Crassostrea virginica	American oyster	Egg	<500	TLM
Nemagon®		Mercenaria mercenaria	Hard clam	Egg	1×10^4	TLM
Nemagon®		Mercenaria mercenaria	Hard clam	Larvae	780	TLM
Nitrofurazone		Mercenaria mercenaria	Hard clam	Egg	$>1\times10^5$	TLM
Nitrofurazone		Mercenaria mercenaria	Hard clam	Larvae	$>1\times10^5$	TLM
Omazene		Mercenaria mercenaria	Hard clam	Egg	81	TLM
Omazene		Mercenaria mercenaria	Hard clam	Larvae	378	TLM
Omazene		Crassostrea virginica	American oyster	Egg	78	TLM
Omazene		Crassostrea virginica	American oyster	Larvae	340	TLM
Nitrilotriacetic acid	Monohydrated sodium salt	Cyclotella nana			5×10^3	38 percent growth as compared to controls
Nitrilotriacetic acid	Monohydrated sodium salt	Tisbe furcata			2.7×10^5	TL-50
Nitrilotriacetic acid	Monohydrated sodium salt	Acartia clausi			1.35×10^6	TL-50
Nitrilotriacetic acid	Monohydrated sodium salt	Trigriopus japonicus			3.2×10^6	TL-50
Nitrilotriacetic acid	Monohydrated sodium salt	Pseudodiaptimus coronatus			7×10^5	TL-50
Nitrilotriacetic acid	Monohydrated sodium salt	Eurytemora affinis			1.25×10^6	TL-50
Nitrilotriacetic acid	Monohydrated sodium salt		Crab zoea		1.65×10^6	TL-50
Nitrilotriacetic acid	Monohydrated sodium salt	Nereis virens	Sand worm	Adult	5.5×10^6	TL-50

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[a] No growth but organisms viable.

Continued

Test Procedure	Temp C	Salinity ‰	Other Environmental Criteria	Statistical Evaluation	Residue levels mg/kg	Other Parameters	Reference
10 day growth test	20.5±1		500 ft.-c continuous	None			Ukeles 1962[347]
10 day growth test	20.5±1		500 ft.-c continuous	None			Ukeles 1962[347]
10 day growth test	20	30	pH=7.9–8.1 6000 lux 12/12	None			Walsh and Grow 1971[349]
[f]	20	30	pH=7.9–8.1	Litchfield & Wilcoxon method[337]			Walsh 1972[348]
10 day growth test	20	30	pH=7.9–8.1 6000 lux 12/12	"			Walsh "
[f]	20	30	pH=7.9–8.1	"			Walsh "
10 day growth test	20	30	pH=7.9–8.1 6000 lux 12/12	"			Walsh "
10 day growth test	20.5±1		500 ft.-c continuous	None			Ukeles 1962[347]
[f]	20	30	pH=7.9–8.1	Litchfield & Wilcoxon method[337]			Walsh 1972[348]
10 day growth test	20	30	pH=7.9–8.1 6000 lux 12/12[d]	"			Walsh "
10 day growth test	20.5±1		500 ft.-c continuous	None			Ukeles 1962[347]
10 day growth test	20.5±1		500 ft.-c continuous	None			Ukeles 1962[347]
[f]	20	30	pH=7.9–8.1	Litchfield & Wilcoxon Method[337]			Walsh 1972[348]
10 day growth test	20	30	pH=7.9–8.1 6000 lux 12/12	None			Walsh "
96 hr static lab bioassay	26	0	Grown in Tetrahymena broth	Decrease significant at 0.05 level			Cooley and Keltner (unpublished)[350]
15 day chronic exposure in flowing sea-water	29	32		Significant at .005 level			Nimmo et al. (unpublished)[355]
35 day chronic exposure in flowing sea-water	20	28		Significant at 0.001 level			Nimmo et al. (unpublished)[355]
18 day chronic exposure in flowing sea-water	11–18	16–32		Significant at 0.05 level	46 ppm		Hansen et al. 1971[332]
12 day chronic exposure in flowing sea-water	16–22	20–32			13 ppm		Hansen et al. 1971[332]
48 hr static lab bioassay	24±1			None			Davis and Hidu 1969[324]
12 day static lab bioassay	24±1			None			Davis and Hidu 1969[324]
12 day static lab bioassay	24±1			None			Davis and Hidu 1969[324]
14 day static lab bioassay	24±1			None			Davis and Hidu 1969[324]
10 day growth test	20.5±1	22–28		None			Ukeles 1962[347]
10 day growth test	20.5±1	22–28		None			Ukeles 1962[347]
10 day growth test	20.5±1	22–28		None			Ukeles 1962[347]
10 day growth test	20.5±1	22–28		None			Ukeles 1962[347]
10 day growth test	20.5±1	22–28		None			Ukeles 1962[347]
48 hr static lab bioassay	24±1			None			Davis and Hidu 1969[324]
12 day static lab bioassay	24±1			None			Davis and Hidu 1969[324]
48 hr static lab bioassay	24±1			None			Davis and Hidu 1969[324]
12 day static lab bioassay	24±1			None			Davis and Hidu 1969[324]
48 hr static lab bioassay	24±1			None			Davis and Hidu 1969[324]
14 day static lab bioassay	24±1			None			Davis and Hidu 1969[324]
10 day growth test	20.5±1	22–28		None			Ukeles 1962[347]
10 day growth test	20.5±1	22–28		None			Ukeles 1962[347]
10 day growth test	20.5±1	22–28		None			Ukeles 1962[347]
10 day growth test	20.5±1	22–28		None			Ukeles 1962[347]
10 day growth test	20.5±1	22–28		None			Ukeles 1962[347]
10 day growth test	20.5±1	22–28		None			Ukeles 1962[347]
10 day growth test	20.5±1	22–28		None			Ukeles 1962[347]
10 day growth test	20.5±1	22–28		None			Ukeles 1962[347]
10 day growth test	20.5±1	22–28		None			Ukeles 1962[347]
10 day growth test	20.5±1	22–28		None			Ukeles 1962[347]
48 hr static lab bioassay	24±1			None			Davis and Hidu 1969[324]
12 day static lab bioassay	24±1			None			Davis and Hidu 1969[324]
48 hr static lab bioassay	24±1			None			Davis and Hidu 1969[324]
48 hr static lab bioassay	24±1			None			Davis and Hidu 1969[324]
12 day static lab bioassay	24±1			None			Davis and Hidu 1969[324]
48 hr static lab bioassay	24±1			None			Davis and Hidu 1969[324]
12 day static lab bioassay	24±1			None			Davis and Hidu 1969[324]
48 hr static lab bioassay	24±1			None			Davis and Hidu 1969[324]
12 day static lab bioassay	24±1			None			Davis and Hidu 1969[324]
48 hr static lab bioassay	24±1			None			Davis and Hidu 1969[324]
14 day static lab bioassay	24±1			None			Davis and Hidu 1969[324]
72 hr static lab bioassay	20	32	250 ft-c 14 hrs on/10 hrs off				Erickson et al. 1970[331]
72 hr static lab bioassay	15(T)	30					NMWQL 1970[341]
72 hr static lab bioassay	15(T)	30					NMWQL 1970[341]
72 hr static lab bioassay	15(T)	30					NMWQL 1970[341]
72 hr static lab bioassay	15(T)	30					NMWQL 1970[341]
72 hr static lab bioassay	15(T)	30					NMWQL 1970[341]
72 hr static lab bioassay	15(T)	30					NMWQL 1970[341]
96 hr static lab bioassay	20	20					NMWQL 1970[344]

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[f] O_2 evolution measured with a Gilson differential respirometer on 4 ml of culture in log-phase.

TABLE 6—

Substance Tested	Formulation	Organism Tested	Common Name	Life Stage or Size (mm)	Conc. (ppb act. ingred.) in water	Methods of Assessment
Nitrilotriacetic acid	Monohydrated sodium salt	Nereis virens	Sand worm	Adult	5.5×10^6	TL-50
Nitrilotriacetic acid	Monohydrated sodium salt	Palaemonetes vulgaris	Grass shrimp	Adult	4.1×10^6	TL-50
Nitrilotriacetic acid	Monohydrated sodium salt	Palaemonetes vulgaris	Grass shrimp	Adult	1.8×10^6	TL-50
Nitrilotriacetic acid	Monohydrated sodium salt	Palaemonetes vulgaris	Grass shrimp		1.0×10^6	subjected to histopathologic examination
Nitrilotriacetic acid	Monohydrated sodium salt	Penaeus setiferus	White shrimp	Sub-adult	1×10^6	78 percent mortality
Nitrilotriacetic acid	Monohydrated sodium salt	Penaeus setiferus	White shrimp	Sub-adult	5×10^6	90 percent mortality
Nitrilotriacetic acid	Monohydrated sodium salt	Homarus americanus	American lobster	Sub-adult (292 grams)	3.8×10^6	TL-50
Nitrilotriacetic acid	Monohydrated sodium salt	Homarus americanus	American lobster	Sub-adult (292 grams)	3.15×10^6	TL-50
Nitrilotriacetic acid	Monohydrated sodium salt	Homarus americanus	American lobster	First larval stage	1×10^5	100 percent mortality
Nitrilotriacetic acid	Monohydrated sodium salt	Uca pugilator	Fiddler crab	Adult	1×10^7	25 percent mortality
Nitrilotriacetic acid	Monohydrated sodium salt	Uca pugilator	Fiddler crab	Adult	1×10^6	46 percent mortality
Nitrilotriacetic acid	Monohydrated sodium salt	Pagurus longicarpus	Hermit crab	Adult	5.5×10^6	TL-50
Nitrilotriacetic acid	Monohydrated sodium salt	Pagurus longicarpus	Hermit crab	Adult	1.8×10^6	TL-50
Nitrilotriacetic acid	Monohydrated sodium salt		Oyster	Larvae	3.5×10^5	46 percent mortality
Nitrilotriacetic acid	Monohydrated sodium salt	Nassa obsoleta	Mud snail	Adult	5.5×10^6	TL-50
Nitrilotriacetic acid	Monohydrated sodium salt	Nassa obsoleta	Mud snail	Adult	5.1×10^6	TL-50
Nitrilotriacetic acid	Monohydrated sodium salt	Mytilus edulis	Bay mussel	Adult	6.1×10^6	TL-50
Nitrilotriacetic acid	Monohydrated sodium salt	Mytilus edulis	Bay mussel	Adult	3.4×10^6	TL-50
Nitrilotriacetic acid	Monohydrated sodium salt	Mercenaria mercenaria	Hard clam	Adult	$>1\times10^7$	TL-50
Nitrilotriacetic acid	Monohydrated sodium salt	Mercenaria mercenaria	Hard clam	Adult	$>1\times10^7$	TL-50
Nitrilotriacetic acid	Monohydrated sodium salt	Asterias forbesi	Starfish	Sub-adult	3×10^6	TL-50
Nitrilotriacetic acid	Monohydrated sodium salt	Asterias forbesi	Starfish	Sub-adult	3×10^6	TL-50
Nitrilotriacetic acid	Monohydrated sodium salt	Fundulus heteroclitus	Mummichog	Adult	5.5×10^6	TL-50
Nitrilotriacetic acid	Monohydrated sodium salt	Fundulus heteroclitus	Mummichog	Adult	5.5×10^6	TL-50
Nitrilotriacetic acid	Monohydrated sodium salt	Fundulus heteroclitus	Mummichog	Adult	1×10^3	Examined for histopathology
Nitrilotriacetic acid	Monohydrated sodium salt	Stenotomus chrysops	Scup	Sub-adult	3.15×10^6	TL-50
Nitrilotriacetic acid	Monohydrated sodium salt	Stenotomus chrysops	Scup	Sub-adult	3.15×10^6	TL-50
Nitrilotriacetic acid	Monohydrated sodium salt	Roccus saxatilis	Striped bass	Juvenile (65 mm)	5.5×10^6	TL-50
Nitrilotriacetic acid	Monohydrated sodium salt	Roccus saxatilis	Striped bass	Juvenile (65 mm)	5.5×10^6	TL-50
Nitrilotriacetic acid	Monohydrated sodium salt	Roccus saxatilis	Striped bass	Juvenile (65 mm)	3×10^6	TL-100, Histopathology
Nitrilotriacetic acid	Monohydrated sodium salt	Roccus saxatilis	Striped bass	Juvenile (65 mm)	10×10^6	TL-0
Phenol		Protococcus sp.			3×10^5	.59 O.D. expt/O.D. control
Phenol		Chlorella sp.			3×10^5	.63 O.D. expt/O.D. control
Phenol		Dunaliella euchlora			1×10^5	.51 O.D. expt/O.D. control
Phenol		Phaeodactylum tricornutum			1×10^5	[A] .00 O.D. expt/O.D. control
Phenol		Monochrysis lutheri			1×10^5	[A] .00 O.D. expt/O.D. control
Phenol		Crassostrea virginica	American oyster	Egg	5.825×10^4	TLM
Phenol		Mercenaria mercenaria	Hard clam	Egg	5.263×10^4	TLM
Phenol		Mercenaria mercenaria	Hard clam	Larvae	5.5×10^4	TLM
Phygon®		Mercenaria mercenaria	Hard clam	Egg	14	TLM
Phygon®		Mercenaria mercenaria	Hard clam	Larvae	1.75×10^3	TLM
Phygon®		Crassostrea virginica	American oyster	Egg	14	TLM
Phygon®		Crassostrea virginica	American oyster	Larvae	41	TLM
PVP-Iodine		Protococcus sp.			$1.\times10^5$	.59 O.D. expt/O.D. control
PVP-Iodine		Chlorella sp.			2×10^4	.65 O.D. expt/O.D. control
PVP-Iodine		Dunaliella euchlora			5×10^4	[A] .00 O.D. expt/O.D. control
PVP-Iodine		Phaeodactylum tricornutum			5×10^4	[A] .00 O.D. expt/O.D. control
PVP-Iodine		Monochrysis lutheri			5×10^4	.61 O.D. expt/O.D. control

[A] No growth but organisms viable.

Continued

Test Procedure	Temp C	Salinity ‰	Other Environmental Criteria	Statistical Evaluation	Residue levels mg/kg	Other Parameters	Reference
168 hr static lab bioassay	20	20	Subdued natural light D.O. ca 4 mg/l; pH 7.8				NMWQL 1970[344]
96 hr static lab bioassay	20	20					NMWQL 1970[344]
168 hr static lab bioassay	20	20	Subdued natural light D.O. ca 4 mg/l; pH 7.8				NMWQL 1970[344]
168 hr static lab bioassay	20	20				Digestive diverticulata histopathology	NMWQL 1970[344]
22 day chronic flowing lab bioassay	18–24	30					NMWQL 1970[344]
96 hr static lab bioassay	20	30					NMWQL 1970[344]
96 hr static lab bioassay	20	20	Subdued natural light D.O. ca 4 mg/l; pH 7.8				NMWQL 1970[344]
168 hr static lab bioassay	20	20	Subdued natural light D.O. ca 4 mg/l; pH 7.8				NMWQL 1970[344]
7 day static lab bioassay	20 ambient	20					NMWQL 1970[344]
96 hr static lab bioassay	20 ambient	30					NMWQL 1970[334]
45 day chronic flowing lab bioassay	18–24	24–30					NMWQL 1970[334]
96 hr static lab bioassay	20	20					NMWQL 1970[334]
168 hr static lab bioassay	20	20					NMWQL 1970[334]
24 hr static lab bioassay							NMWQL 1970[334]
96 hr static lab bioassay	20	20	Subdued natural light D.O. ca 4 mg/l; pH 7.8				NMWQL 1970[334]
168 hr static lab bioassay	20	20	Subdued natural light D.O. ca 4 mg/l; pH 7.8				NMWQL 1970[334]
96 hr static lab bioassay	20	20	Subdued natural light D.O. ca 4 mg/l; pH 7.8				NMWQL 1970[334]
168 hr static lab bioassay	20	20	Subdued natural light D.O. ca 4 mg/l pH 7.8				NMWQL 1970[334]
96 hr static lab bioassay	20	20	Subdued natural light D.O. ca 4 mg/l pH 7.8				NMWQL 1970[334]
168 hr static lab bioassay	20	20	Subdued natural light D.O. ca 4 mg/l pH 7.8				NMWQL 1970[334]
96 hr static lab bioassay	20	20	Subdued natural light D.O. ca 4 mg/l; pH 7.8				NMWQL 1970[334]
168 hr static lab bioassay	20	20	Subdued natural light D.O. ca 4 mg/l; pH 7.8				NMWQL 1970[334]
96 hr static lab bioassay	20	20	Subdued natural light D.O. ca 4 mg/l; pH 7.8				NMWQL 1970[334]
168 hr static lab bioassay	20	20	Subdued natural light D.O. ca 4 mg/l; pH 7.8				NMWQL 1970[334]
168 hr static lab bioassay	20	20	Subdued natural light D.O. ca 4 mg/l; pH 7.8			Intestinal Pathology	NMWQL 1970[334]
96 hr static lab bioassay	20	20	Subdued natural light D O ca 4 mg/l; pH 7.8				NMWQL 1970[334]
168 hr static lab bioassay	20	20	Subdued natural light D.O. ca 4 mg/l; pH 7.8				NMWQL 1970[334]
96 hr static lab bioassay	20	20	Subdued natural light D.O. ca 4 mg/l; pH 7.8				NMWQL 1970[334]
168 hr static lab bioassay	20	20	Subdued natural light D.O. ca 4 mg/l; pH 7.8				NMWQL 1970[334]
168 hr static lab bioassay	20	20	Subdued natural light D.O. ca 4 mg/l; pH 7.8			Renal Pathology	NMWQL 1970[334]
168 hr static lab bioassay	20	20	Subdued natural light D.O. ca 4 mg/l; pH 7.8				NMWQL 1970[334]
10 day growth test	20.5±1	22–28		None			Ukeles 1962[347]
10 day growth test	20.5±1	22–28		None			Ukeles 1962[347]
10 day growth test	20.5±1	22–28		None			Ukeles 1962[347]
10 day growth test	20.5±1	22–28		None			Ukeles 1962[347]
10 day growth test	20.5±1	22–28		None			Ukeles 1962[347]
48 hr static lab bioassay	24±1			None			Davis and Hidu 1969[324]
48 hr static lab bioassay	24±1			None			Davis and Hidu 1969[324]
12 day static lab bioassay	24±1			None			Davis and Hidu 1969[324]
48 hr static lab bioassay	24±1			None			Davis and Hidu 1969[324]
12 day static lab bioassay	24±1			None			Davis and Hidu 1969[324]
48 hr static lab bioassay	24±1			None			Davis and Hidu 1969[324]
14 day static lab bioassay	24±1			None			Davis and Hidu 1969[324]
10 day growth test	20.5±1	22–28		None			Ukeles 1962[347]
10 day growth test	20.5±1	22–28		None			Ukeles 1962[347]
10 day growth test	20.5±1	22–28		None			Ukeles 1962[347]
10 day growth test	20.5±1	22–28		None			Ukeles 1962[347]
10 day growth test	20.5±1	22–28		None			Ukeles 1962[347]

TABLE 6—

Substance Tested	Formulation	Organism Tested	Common Name	Life Stage or Size (mm)	Conc. (ppb act. ingred.) in water	Methods of Assessment
PVP-Iodine		Mercenaria mercenaria	Hard clam	Egg	1.71×10^4	TLM
PVP-Iodine		Mercenaria mercenaria	Hard clam	Larvae	3.494×10^4	TLM
Roccal®		Mercenaria mercenaria	Hard clam	Egg	190	TLM
Roccal®		Mercenaria mercenaria	Hard clam	Larvae	140	TLM
Sulmet	Tinted	Mercenaria mercenaria	Hard clam	Egg	1×10^5	TLM
Sulmet	Tinted	Mercenaria mercenaria	Hard clam	Larvae	1×10^5	TLM
TCC		Mercenaria mercenaria	Hard clam	Egg	32	TLM
TCC		Mercenaria mercenaria	Hard clam	Larvae	37	TLM
TCP		Crassostrea virginica	American oyster	Egg	600	TLM
TCP		Crassostrea virginica	American oyster	Larvae	1×10^3	TLM

Continued

Test Procedure	Temp C	Salinity °/₀₀	Other Environmental Criteria	Statistical Evaluation	Residue levels mg/kg	Other Parameters	Reference
48 hr static lab bioassay	24±1	....	....	None	....	....	Davis and Hidu 1969[524]
12 day static lab bioassay	24±1	....	....	None	....	....	Davis and Hidu 1969[524]
48 hr static lab bioassay	24±1	....	....	None	....	....	Davis and Hidu 1969[524]
12 day static lab bioassay	24±1	....	....	None	....	....	Davis and Hidu 1969[524]
48 hr static lab bioassay	24±1	....	....	None	....	....	Davis and Hidu 1969[524]
12 day static lab bioassay	24±1	....	....	None	....	....	Davis and Hidu 1969[524]
48 hr static lab bioassay	24±1	....	....	None	....	....	Davis and Hidu 1969[524]
12 day static lab bioassay	24±1	....	....	None	....	....	Davis and Hidu 1969[524]
48 hr static lab bioassay	24±1	....	....	None	....	....	Davis and Hidu 1969[524]
14 day static lab bioassay	24±1	....	....	None	....	....	Davis and Hidu 1969[524]

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GLOSSARY

absorption penetration of one substance into the body of another.

acclimation the process of adjusting to change, e.g. temperatures, in an environment.

acute involving a stimulus severe enough to rapidly induce a response; in bioassay tests, a response observed within 96 hours typically is considered an acute one.

adsorption the taking up of one substance at the surface of another.

aerobic the condition associated with the presence of free oxygen in an environment.

aerobe an organism that can live and grow only in the presence of free oxygen.

allocthanous said of food material reaching an aquatic community from the outside in the form of organic detritus.

alluvial transported and deposited by running water.

amoebiasis an infection caused by amoebas, especially by *Entamoeba histolytica*.

amphoteric able to react as either acid or base.

anadromous fish fish that typically inhabit seas or lakes but ascend streams at more or less regular intervals to spawn; e.g., salmon, steelhead, or American shad.

anaerobic the condition associated with the lack of free oxygen in an environment.

anaerobe an organism for whose life processes a complete or nearly complete absence of oxygen is essential.

anhydremia a deficiency of water in the blood.

anorexia loss of appetite.

anoxic depleted of free oxygen; anaerobic.

antagonism the power of one toxic substance to diminish or eliminate the toxic effect of another; interactions of organisms growing in close association, to the detriment of at least one of them.

application factor a factor applied to a short-term or acute toxicity test to estimate a concentration of waste that would be safe in a receiving water.

assimilation the transformation and incorporation of substances (e.g., nutrients) by an organism or ecosystem.

backwashing cleaning a filter or ion exchanger by reversing the flow of liquid through it and washing out captured matter.

benthic aquatic bottom-dwelling organisms including: (1) sessile animals, such as the sponges, barnacles, mussels, oysters, some worms, and many attached algae; (2) creeping forms, such as insects, snails, and certain clams; and (3) burrowing forms which include most clams and worms.

bioaccumulation uptake and retention of environmental substances by an organism from its environment, as opposed to uptake from its food.

bioassay a determination of the concentration or dose of a given material necessary to affect a test organism under stated conditions.

biomass the living weight of a plant or animal population, usually expressed on a unit area basis.

biotic index a numerical index using various aquatic organisms to determine their degree of tolerance to differing water conditions.

biotoxin toxin produced by a living organism; the biotoxin which causes paralytic shellfish poisoning is produced by certain species of dinoflagellate algae.

black liquor waste liquid remaining after digestion of rags, straw, and pulp.

bloom an unusually large number of organisms per unit of water, usually algae, made up of one or a few species; a bur of iron or steel, square or slightly oblong, rolled from an ingot to a size intermediate between an ingot and a billet, generally in the range of 6″×6″ to 10″×12″ (Section VI).

blowdown the discharge of water from a boiler or cooling tower to dispose of accumulated salts.

body burden the total amount of a substance present in the body tissues and fluids of an organism.

boiler feedwater water provided to a boiler for conversion to steam in the steam generation process; usually a mixture of make-up water and returned steam condensate.

buffer capacity the ability of a solution to maintain its pH when stressed chemically.

capillary water the water held in the small pores of a soil, usually with a tension greater than 60 centimeters of water.

carrying capacity the maximum biomass that a system is capable of supporting continuously (Section IV); the number of user-use periods that a recreation resource can provide in a given time span without appreciable biological or physical deterioration of that resource, or without appreciable impairment of the recreation experience for the majority of the users (Section I).

catadromous fish fishes that feed and grow in fresh water but return to the sea to spawn, e.g., the American eel.

chelate to combine with a metal ion and hold it in solution preventing it from forming an insoluble salt.

chemotaxis orientation or movement of a living organism in response to a chemical gradient.

chronic involving a stimulus that lingers or continues for a long period of time, often one-tenth of the life span or more.

climax community the stage of ecological development at which a community becomes stable, self-perpetuating, and at equilibrium with the environment.

coagulation a water treatment process in which chemicals are added to combine with or trap suspended and colloidal particles to form rapidly settling aggregates.

coliform bacteria a group of bacteria inhabiting the intestines of animals including man, but also found elsewhere. It includes all the aerobic, nonspore forming, rod-shaped bacteria that produce from lactose fermentation within 48 hours at 37 C.

colloid very small particles (10 anistroms to 1 micron) which tend to remain suspended and dispersed in liquids.

colluvial material that has moved down hill by the force of gravity or frost action and local wash and accumulated on lower slopes or at the bottom of the hill.

conjunctivitis an inflammation of the mucous membrane that lines the inner surface of the eyelid and the exposed surface of the eyeball.

conservative pollutant a pollutant that is relatively persistant and resistant to degradation, such as PCB and most chlorinated hydrocarbon insecticides.

cumulative brought about or increased in strength by successive additions.

demersal living or hatching on the bottom, as fish eggs than sink to the bottom.

detritus unconsolidated sediments comprised of both inorganic and dead and decaying organic material.

diurnal occurring once a day, i.e., with a variation period of 1 day; occurring in the daytime or during a day.

diversity the abundance in numbers of species in a specified location.

dredge spoils the material removed from the bottom during dredging operations.

drench to administer orally with water a large dose of a substance such as medicine to an animal.

dystrophic said of brownwater lakes and streams usually with a low lime content and a high organic content; often lacking in nutrients.

emesis the act of vomiting.

enteric of or originating in the intestinal tract.

epilimnion the surface waters in a thermally stratified body of water; these waters are characteristically well mixed.

epiphytic living on the surface of other plants.

euphotic zone the lighted region that extends vertically from the water surface to the level at which photosynthesis fails to occur because of ineffective light penetration.

eutrophic abundant in nutrients and having high rates of productivity frequently resulting in oxygen depletion below the surface layer.

evapotranspiration the combined loss of water from a given area during a specified period of time by evaporation from the soil or water surface and by transpiration from plants.

exchange capacity the total ionic charge of the adsorption complex active in the adsorption of ions.

exophthalmos an abnormal protrusion of the eyeball.

external treatment passage of water through equipment such as a filter or water softener to meet desired quality requirements prior to point of use.

facultative able to live under different conditions, as in facultative aerobes and facultative anaerobes.

fecal coliform bacteria bacteria of the coliform group of fecal origin (from intestines of warm-blooded animals) as opposed to coliforms from non-fecal sources.

filial generation the offspring of a cross mating.

finfish that portion of the aquatic community made up of the true fishes as opposed to invertebrate shellfish.

flocculation the process by which suspended colloidal or very fine particles are assembled into larger masses or floccules which eventually settle out of suspension; the stirring of water after coagulant chemicals have been added to promote the formation of particles that will settle (Section II).

food chain the transfer of food energy from plants or organic detritus through a series of organisms, usually four or five, consuming and being consumed.

food web the interlocking pattern formed by a series of interconnecting food chains.

free residual chlorination chlorination that maintains the presence of hypochlorous acid (HOCl) or hypochlorite ion (OCl^-) in water.

fry the stage in the life of a fish between the hatching of the egg and the absorption of the yolk sac (Sections

III and IV); in a broader sense, all immature stages of fishes.

groundwood the raw material produced from both logs and chips, used mainly in the manufacture of newsprint, toweling, tissue, wallpaper, and coated specialty papers.

half-life the period of time in which a substance loses half of its active characteristics (used especially in radiological work); the time required to reduce the concentration of a material by half.

hemostasis the cessation of the flow of blood in the circulatory system.

histopathologic occurring in tissue due to a diseased condition.

hydrophobic unable to combine with or dissolve in water.

hydrophytic growing in or in close proximity to water; e.g., aquatic algae and emergent aquatic vascular plants.

hypertrophy nontumorous increase in the size of an organ as a result of enlargement of constituent cells without an increase in their number.

hypolimnion the region of a body of water that extends from below the thermocline to the bottom of the lake; it is thus removed from much of the surface influence.

internal treatment treating water by addition of chemicals to meet desired quality requirements at point of use or within a process.

intraperitoneal into the abdominal cavity.

kraft process producing pulp from wood chips in the manufacture of paper products; involves cooking the chips in a strong solution of caustic soda and sodium sulfide.

labile unstable and likely to change under certain influences.

LC50 see median lethal concentration.

LD50 see median lethal dose.

lentic or lenitic environment standing water and its various intergrades; e.g., lakes, ponds, and swamps.

leptospirosis a disease of animals or man caused by infection from an organism of the genus *Leptospira*.

lethal involving a stimulus or effect causing death directly.

life cycle the series of life stages in the form and mode of life of an organism, i.e., between successive recurrences of a certain primary stage such as the spore, fertilized egg, seed, or resting cell.

limnetic zone the open-water region of a lake, supporting plankton and fish as the principal plants and animals.

lipophilic having an affinity for fats or other lipids.

littoral zone the shallow shoreward region of a body of water having light penetration to the bottom; frequently occupied by rooted plants.

littoral zone the shoreward or coastal region of a body of water.

lotic environment running waters, such as streams or rivers.

lysimeter a device to measure the quantity or rate of water movement through or from a block of soil, usually undisturbed and in situ, or to collect such percolated water for quality analysis.

macronutrient a chemical element necessary in large amounts, usually greater than 1 ppm, for the growth and development of plants.

macrophyte the larger aquatic plants, as distinct from the microscopic plants, including aquatic mosses, liverworts and larger algae as well as vascular plants; no precise toxonomic meaning; generally used synonymously with aquatic vascular plants in this Report.

make-up water water added to boiler, cooling tower, or other systems to maintain the volume of water required.

marl an earthy, unconsolidated deposit formed in freshwater lakes, consisting chiefly of calcium carbonate mixed with clay or other impurities in varying proportions.

median lethal concentration (LC50) the concentration of a test material that causes death to 50 per cent of a population within a given time period.

median lethal dose (LD50) the dose of a test material, ingested or injected, that kills 50 per cent of a group of test organisms.

median tolerance limit (TL50) the concentration of a test material in a suitable diluent (experimental water) at which just 50 per cent of the test animals are able to survive for a specified period of exposure.

mercerize to treat cotton thread with sodium hydroxide so as to shrink the fiber and increase its color absorption and luster.

mesotrophic having a nutrient load resulting in moderate productivity.

metabolites products of metabolic processes.

methemoglobinemia poisoning resulting from the oxidation of ferrous iron of hemoglobin to the ferric state where it is unable to combine reversibly with molecular oxygen; agents responsible include chlorates, nitrates, ferricyanides, sulfonamides, salicylates, and various other substances.

methylation combination with the methyl radical (CH_3).

mho a unit of conductance reciprocal to the ohm

micelle an aggregation or cluster of molecules, ions, or minute submicroscopic particles.

micronutrient chemical element necessary in only small amounts for growth and development; also known as trace elements.

mouse unit the amount of paralytic shellfish poison that will produce a mean death time of 15 minutes when administered intraperitoneally to male mice, of a specific strain, weighing between 18 and 20 grams.

necrosis the death of cellular material within the body of an organism.

nephrosclerosis a hardening of the tissues of the kidney.

nitrilotriacetate (NTA) the salt of nitrilotriacetic acid; has the ability to complex metal ions, and has been proposed as a builder for detergents.

nonconservative pollutant a pollutant that is quickly degraded and lacks persistence, such as most organophosphate insecticides.

nonfouling a property of cooling water that allows it to flow over steam condenser surfaces without accumulation of impediments.

nonpolar a chemical term for any molecule or liquid that has a reasonable degree of electrical symmetry such that there is little or no separation of charge; e.g., benzene, carbon tetrachloride, and the lower paraffin hydrocarbons.

nutrients organic and inorganic chemicals necessary for the growth and reproduction of organisms.

oligotrophic having a small supply of nutrients and thus supporting little organic production, and seldom if every becoming depleted of oxygen.

organoleptic pertaining to or perceived by a sensory organ.

parr a young fish, usually a salmonid, between the larval stage and the time it begins migration to the sea.

partition coefficient the ratio of the molecular concentration of a substance in two solvents.

pCi—picocurie a measure of radioactivity equivalent to 3.70×10^{-2} atoms disintegrating per minute.

pelagic occurring or living in the open ocean.

periphyton associated aquatic organisms attached or clinging to stems and leaves of rooted plants or other surfaces projecting above the bottom of a water body.

pesticide any substance used to kill plants, insects, algae, fungi, and other organisms; includes herbicides, insecticides, algalcides, fungicides, and other substances.

plankton plants (phytoplankton) and animals (zooplankton), usually microscopic, floating in aquatic systems such as rivers, ponds, lakes, and seas.

point of supply the location at which water is obtained from a specific source.

point of use the location at which water is actually used in a process or incorporated into a product.

prime to cause an explosive evolution of steam from a heating surface, throwing water into a steam space.

process water water that comes in contact with an end product or with materials incorporated in an end product.

productivity the rate of storage of organic matter in tissue by organisms including that used by the organisms in maintaining themselves.

pycnocline a layer of water that exhibits rapid change in density, analogous to thermocline.

psychrophilic thriving at relatively low temperatures, usually at or below 15 C.

recharge to add water to the zone of saturation, as in recharge of an aquifer; the term may also be applied to the water added.

refractory resisting ordinary treatment and difficult to degrade.

rip-rapping covering stream banks and dam faces with rock or other material to prevent erosion from water contact.

safety factor a numerical value applied to short-term data from other organisms in order to approximate the concentration of a substance that will not harm or impair the organism being considered.

sessile organism motionless organisms that reside in a fixed state, attached or unattached to a substrate.

seston suspended particles and organisms between 0.0002 and 1 mm in diameter.

shellfish a group of mollusks usually enclosed in a self-secreted shell; includes oysters and clams.

shoal water shallow water.

slaking adding water to lessen the activity of a chemical reaction.

sludge a solid waste fraction precipitated by a water treatment process.

smolt a young fish, usually a salmonid, as it begins and during the time it makes its seaward migration.

sorption a general term for the processes of absorption and adsorption.

species diversity a number which relates the density of organisms of each type present in a habitat.

standing crop biomass the total weight of organisms present at any one time.

stoichiometric the mass relationship in a chemical reaction.

stratification the phenomenon occurring when a body of water becomes divided into distinguishable layers.

subacute involving a stimulus not severe enough to bring about a response speedily.

sublethal involving a stimulus below the level that causes death.

succession the orderly process of community change in which a sequence of communities replaces one another in a given area until a climax community is reached.

sulfhemoglobin the reaction product of oxyhemoglobin and hydrogen sulfide.

sullage waste materials or refuse; sewage.

superchlorination chlorination wherein the doses are large enough to complete all chlorination reactions and to produce a free chlorine residual.

surfactant a surface active agent altering the interfacial tension of water and other liquids or solids, e.g. a detergent.

synergistic interactions of two or more substances or organisms producing a result that any was incapable of independently.

tailwater water, in a river, or canal, immediately downstream from a structure; in irrigation, the water that reaches the lower end of a field.

teart a disease of cattle caused by excessive molybdenum intake characterized by profuse scouring, loss of pigmentation of the hair, and bone defects.

teratogen a substance that increases the incidence of birth defects.

thermocline a layer in a thermally stratified body of water in which the temperature changes rapidly relative to the remainder of the body.

TLm see median tolerance limit.

trophic accumulation passing of a substance through a food chain such that each organism retains all or a portion of the amount in its food and eventually acquires a higher concentration in its flesh than in the food.

trophic level a scheme of categorizing organisms by the way they obtain food from primary producers or organic detritus involving the same number of intermediate steps.

true color the color of water resulting from substances which are totally in solution; not to be mistaken for apparent color resulting from colloidal or suspended matter.

CONVERSION FACTORS

Units	Multiplied by	Equal
Acres	4.047×10^{-1}	Hectares
	4.356×10^{4}	Square feet
	4.047×10^{3}	Square meters
	1.562×10^{-3}	Square miles
	4.840×10^{3}	Square yards
Angstrom units	1×10^{-8}	Centimeters
	3.937×10^{-9}	Inches
Barrels (oil)	42	Gallons (oil)
	1.590×10^{2}	Liters
British thermal units	7.776×10^{2}	Foot pounds
	3.927×10^{-4}	Horse-power hours
	0.252	Kilogram calories
	2.929×10^{-4}	Kilowatt hours
Centimeters	3.937×10^{-1}	Inches
Degrees centigrade	$(°C\times\frac{9}{5})+32$	Farenheit degrees
Degrees farenheit	$(°F-32)\frac{5}{9}$	Centigrade degrees
Feet	12	Inches
	1.646×10^{-4}	Miles (nautical)
	1.894×10^{-4}	Miles (statute)
	0.305	Meters
	1/3	Yards
Gallons	3.069×10^{-6}	Acre feet
	3.785×10^{3}	Cubic centimeters
	0.134	Cubic feet
	2.31×10^{2}	Cubic inches
	3.785×10^{-3}	Cubic meters
	4.951×10^{-3}	Cubic yards
	3.785	Liters
	8	Pints (liquid)
	4	Quarts (liquid)
(Imperial)	1.201	U. S. gallons
(U. S.)	0.833	Imperial gallons
(Water)	8.345	Pounds (Water: 39.2 F)
Gallons/day	5.570×10^{-3}	Cubic feet/hour
	3.785	Liters/day
Gallons/minute	8.021	Cubic feet/hour
	2.228×10^{-3}	Cubic feet/second
	6.308×10^{-2}	Liters/second
(Water)	6.009	Tons (water: 39.2 F)/day
Gallons/square foot/minute	40.74	Liters/square meter/minute
Gallons/square mile	1.461	Liters/square kilometer
Gallons/ton (short)	4.173	Liters/ton (metric)
Grams	3.527×10^{-2}	Ounces
	2.205×10^{-3}	Pounds
Grams/liter	58.41	Grains/gallon
	10^{3}	Parts per million

(assumes density of 1 gram/milliliter)

CONVERSION FACTORS—*Continued*

Units	Multiplied by	Equal
Grams/liter	8.345×10^{-3}	Pounds/gallon
	6.243×10^{-2}	Pounds/cubic foot
Grams/cubic meter	0.437	Grains/cubic foot
Inches	2.54	Centimeters
Kilograms	2.205	Pounds
	1.102×10^{-3}	Tons (short)
	9.842×10^{-4}	Tons (long)
Kilometers	3.281×10^{3}	Feet
	3.937×10^{4}	Inches
	0.621	Miles (statute)
	0.540	Miles (nautical)
	1.094×10^{3}	Yards
Liters	1.000028×10^{3}	Cubic centimeters
	3.532×10^{-2}	Cubic feet
	61.03	Cubic inches
	1.000028×10^{-3}	Cubic meters
	1.308×10^{-3}	Cubic yards
	0.227	Gallons
Liters/square kilometer	0.588	Gallons/square mile
Meters	3.281	Feet
	39.37	Inches
	5.400×10^{-4}	Miles (nautical)
	6.214×10^{-4}	Miles (statute)
	1.094	Yards
Microns	10^{4}	Angstrom units
	10^{-4}	Centimeters
	3.281×10^{-6}	Feet
	3.937×10^{-5}	Inches
	10^{-6}	Meters
	10^{-3}	Millimeters
Miles (nautical)	6.076×10^{3}	Feet
	1.852	Kilometers
	1.852×10^{3}	Meters
	1.151	Miles (statute)
	2.027×10^{3}	Yards
Miles (statute)	5.280×10^{3}	Feet
	6.336×10^{4}	Inches
	1.609	Kilometers
	1.609×10^{3}	Meters
	0.869	Miles (nautical)
	1.760×10^{3}	Yards
Milligrams	3.527×10^{-5}	Ounces
	2.205×10^{-6}	Pounds
Milliliters	1.000028	Cubic centimeters
	6.102×10^{-2}	Cubic inches
	3.381×10^{-2}	Ounces (U. S.)
Millimeters	3.281×10^{-3}	Feet
	3.937×10^{-2}	Inches
	10^{-3}	Meters
	10^{3}	Microns
	1.094×10^{-3}	Yards

CONVERSION FACTORS—*Continued*

Units	Multiplied by	Equal
Million gallons/day	1.547	Cubic feet/second
	0.028	Cubic meters/second
	28.32	Liters/second
Pounds	0.454	Kilograms
	16	Ounces
	4.464×10^{-4}	Tons (long)
	4.536×10^{-4}	Tons (metric)
	5.0×10^{-4}	Tons (short)
Pounds/acre	1.122	Kilograms/hectare
Pounds/gallon	0.120	Grams/cubic centimeter
	7.480	Pounds/cubic foot
Pounds/square inch	6.805×10^{-2}	Atmospheres
	5.171	Centimeters of mercury (0 C)
	70.31	Centimeters of water (4 C)
	6.895×10^{4}	Dynes/square centimeter
	70.31	Grams/square centimeter
	27.68	Inches of water (39.2 F)
	2.036	Inches of mercury (32 F)
	7.031×10^{2}	Kilograms/square meter
	1.440×10^{2}	Pounds/square foot
Square feet	2.296×10^{-5}	Acres
	1.44×10^{2}	Square inches
	9.290×10^{-2}	Square meters
	3.587×10^{-8}	Square miles
	1/9	Square yards
Square meters	2.471×10^{-4}	Acres
	10^{-4}	Hectares
	10^{4}	Square centimeters
	10.76	Square feet
	1.550×10^{3}	Square inches
	3.861×10^{-7}	Square miles
	1.196	Square yards
Square miles	6.40×10^{2}	Acres
	2.590×10^{2}	Hectares
	2.788×10^{7}	Square feet
	2.590	Square kilometers
	3.098×10^{6}	Square yards
Tons (metric)	10^{3}	Kilograms
	3.527×10^{4}	Ounces
	2.205×10^{3}	Pounds
	0.984	Tons (long)
	1.102	Tons (short)
Tons (short)	8.897×10^{8}	Dynes
	9.072×10^{2}	Kilograms
	3.2×10^{4}	Ounces

CONVERSION FACTORS—*Continued*

Units	Multiplied by	Equal
Tons (short)	2×10^3	Pounds
	0.893	Tons (long)
	0.907	Tons (metric)
Watts	3.414	BTU/hour
	44.25	Foot-pounds/minute
	1.341×10^{-3}	Horse power
	1.434×10^{-2}	Kilogram-calories/minute
Yards	91.44	Centimeters
	3	Feet
	36	Inches
	0.914	Meters
	4.934×10^{-4}	Miles (nautical)
	5.682×10^{-4}	Miles (statute)

BIOGRAPHICAL NOTES

Committee on Water Quality Criteria

Gerard A. Rohlich is C. W. Cook Professor of Environmental Engineering and Professor at the Lyndon B. Johnson School of Public Affairs at the University of Texas at Austin. He received B.S. degrees from Cooper Union in 1934 and the University of Wisconsin in 1936, and an M.S. in 1937 and a Ph.D. in Sanitary Engineering in 1940 from the University of Wisconsin. Dr. Rohlich's expertise is in wastewater treatment and in eutrophication and pollution of lakes and streams. He is a member of the National Academy of Engineering.

Alfred M. Beeton is Associate Director (Biology) of the Center for Great Lakes Studies, and Professor of Zoology at the University of Wisconsin, Milwaukee. He received a B.S. degree from the University of Michigan in 1952, an M.S. in 1954 and Ph.D. in Zoology in 1958 from the University of Michigan. Dr. Beeton's major research interest is the eutrophication of the Great Lakes. He is Coordinator of the Water Quality Sub-program of the University of Wisconsin Sea Grant Program.

Bostwick H. Ketchum is Associate Director of the Woods Hole Oceanographic Institution in Woods Hole, Massachusetts. He received an A.B. in Biology from St. Stephens College, Columbia, in 1934, and a Ph.D. from Harvard University in 1938. A specialist in nutrient cycling and phytoplankton physiology, he is the holder of two Honorary Sc.D. degrees and the 1972 recipient of the David B. Stone Award.

Cornelius W. Krusé is Professor and Chairman of the Department of Environmental Health in the School of Hygiene and Public Health at the Johns Hopkins University. He received a B.S. in civil engineering at the Missouri School of Mines in 1934, an M.S. in Sanitary Engineering at Harvard University in 1940, and a Doctor of Public Health from the University of Pittsburgh in 1961. Dr. Krusé is a specialist on infectious and toxic agents in external environments.

Thurston E. Larson received a B.S. in Chemical Engineering in 1932 and a Ph.D. in Sanitary Chemistry in 1937 from the University of Illinois. He is a specialist in water chemistry and head of the Chemistry Section of the Illinois State Water Survey. Dr. Larson is a past-president of the American Water Works Association.

Emilio A. Savinelli is a specialist in water and wastewater treatment and President of Drew Chemical Corpora-Parsippany, New Jersey. He received a B.C.E. in Sanitary Engineering in 1950 from Manhattan College and a M.S.E. in 1951 and a Ph.D. in Chemistry in 1955 from the University of Florida.

Ray L. Shirley received his B.S. and M.S. in agriculture at the University of West Virginia in 1937 and 1939, respectively, and a Ph.D. in agricultural biochemistry from Michigan State University in 1949. Dr. Shirley specializes in animal nutrition and metabolism. He is Professor of Animal Science and in charge of the Nutrition Laboratory at the University of Florida, Gainesville.

Panel On Recreation and Aesthetics

Michael Chubb is Associate Professor of Environmental and Recreational Geography at Michigan State University. He received a B.Sc.F. from the Faculty of Forestry, University of Toronto in 1955, a M.S. in resource development in 1964 and a Ph.D. in Geography from M.S.U. in 1967. He has been employed in river development for the Ontario Department of the Environment and has served as a recreation planner with the Michigan Department of Natural Resources. He is a specialist in recreation resource carrying capacity and recreation survey research.

Milo A. Churchill is Chief, Water Quality Branch, Tennessee Valley Authority. He received a B.S. in Civil Engineering from the University of Illinois in 1933, and a Master of Public Health from The Johns Hopkins University in 1952. He is a specialist in effects of impoundment on water quality, bacterial quality of streams and reservoirs, and stream reaeration.

Norman E. Jackson is Acting Assistant Chief, Planning Division, Office of the Chief of Engineers, Directorate of Civil Works, Washington, D. C. He received a B.S.

in Civil Engineering from Vanderbilt University in 1935 and a M.S. in Sanitary Engineering from The Johns Hopkins University in 1950. A former Director of Sanitary Engineering, District of Columbia, and Special Assistant to the Director, Department of Environmental Services, D.C., he specializes in urban water and wastewater facilities and in planning water resources in regional and basin settings.

William L. Klein is chemist-biologist for the Ohio River Valley Water Sanitation Commission at Cincinnati, Ohio. He received a B.S. degree from Kent State in 1949, and a Master of Public Health in 1957 in Sanitary Chemistry and Biology from the University of North Carolina. He is a specialist in water pollution control, water monitoring, and associated data evaluation.

Percy H. McGauhey is a professional consultant in civil and sanitary engineering and Director Emeritus of Sanitary Engineering Research Laboratory, Department of Civil Engineering, University of California, Berkeley. He received an M.S. degree from the University of Wisconsin in 1941, and an honorary D.Sc. from Utah State University in 1971. He is the author of *Engineering Water Quality Management* (McGraw-Hill, 1968). His specialties are wastewater reclamation, organic clogging of soils, economic evaluation of water, and solid wastes management.

Eric W. Mood is Associate Professor of Public Health (Environmental Health), Yale University School of Medicine. He received a B.S. degree in engineering from the University of Connecticut in 1938 and an M.P.H. degree from Yale University in 1943. For several years he has been chairman of the Joint Committee on Swimming Pools and Bathing Places, American Public Health Association. He is a member of the Expert Advisory Committee on Environmental Health, World Health Organization, and is the recipient of an honorary Doctor of Laws degree from Upsala College.

Ralph Porges is Head of the Water Quality Branch of the Delaware River Basin Commission. He received a B.S. in Chemical Engineering from Rutgers University in 1936 and spent four years as a Research Fellow at the University of North Carolina. He was with the U.S. Public Health Service for 26 years, specializing in stream pollution and plague and typhus control. He is a holder of the William D. Hatfield Award of the Water Pollution Control Federation.

Leslie M. Reid is Professor and Head of the Department of Recreation and Parks at Texas A&M University. He received a B.S. in Forestry from the Michigan Technological University in 1951, an M.S. in Resource Development from Michigan State University in 1955, and a Ph.D. in Conservation in 1963 from the University of Michigan. His major research interest is in natural resources planning. He is consultant with the National Park Service and a past-president of the Society of Park and Recreation Educators.

Michael B. Sonnen is a Senior Engineer with Water Resources Engineers, Inc. in Walnut Creek, California. He received a B.E. in Civil Engineering at Vanderbilt University in 1962, an M.S. in Sanitary Engineering in 1965, and a Ph.D. in Sanitary Engineering in 1967, both from the University of Illinois. His major research interest is the evaluation of costs and benefits accruing to water users supplied with various qualities of water.

Robert O. Sylvester is Professor and Head, Division of Water and Air Resources, Department of Civil Engineering, University of Washington. He received a B.S. in Civil Engineering from Harvard University in 1941. His teaching, research and professional interests have centered around water quality, water supply, and water resources-quality management.

C. William Threinen is Administrative Assistant, Department of Natural Resources, State of Wisconsin, Madison. He is also acting assistant director of the Bureau of Fish Management in the Department of Natural Resources. He received his B.S. degree from the University of Wisconsin, and his M.S. in public administration from Harvard University. He specializes in rough fish problems in Wisconsin, population dynamics of largemouth bass, lake use studies, wild rivers planning, and access site utilization and development.

Panel On Public Water Supplies

Russell F. Christman is Director, Division of Environmental Affairs and Professor of Applied Chemistry at the University of Washington. He received an M.S. in 1960 and a Ph.D. in 1962 in Chemistry from the University of Florida. His major research activities involve the identification of organic materials in natural water systems.

Paul D. Haney is a member of the firm of Black & Veatch, Consulting Engineers, Kansas City, Missouri. He received a B.S. degree in Chemical Engineering from the University of Kansas in 1933, and an M.S. degree in Sanitary Engineering from Harvard University in 1937. He is a diplomate of the American Academy of Environmental Engineers and a past-president of the Water Pollution Control Federation.

Robert C. McWhinnie, Chief of Planning and Resource Development, Board of Water Commissioners, Denver, Colorado. He received a B.S. degree in civil engineering from the University of Wyoming and has taken advanced courses in sanitary engineering from the University of Colorado. He is Director of the Metropolitan Denver Sewage Disposal District and chairman of the Technical Advisory Committee on Urban Flood Control. He specializes in odor, temperature, and phenolic compounds in public water supplies.

Henry J. Ongerth is Chief of the Bureau of Sanitary Engineering, State Department of Public Health, California. He received a B.S. degree from University of California (Berkeley) in civil engineering (sanitary option) in 1935 and an M.P.H. in 1950 from the University of Michigan. He was a member of the advisory committee of the 1962 revision of the Public Health Service Drinking Water Standards and chairman of the Public Advisory Committee presently working on the 1972 revision of the Drinking Water Standards. He is a member of the American Academy of Environmental Engineers and a specialist in fluorides, chlorides, and sulfates in water.

Ranard J. Pickering is Chief of the Quality of Water Branch, Water Resources Division, U.S. Geological Survey at Washington, D.C. He received an A.B. with Highest Honors in 1951 and an M.A. in 1952 from Indiana University, and a Ph.D. from Stanford University in 1961, all in Geology. His specialties are low temperature geochemistry and behavior of radionuclides in fluvial environments.

Josehp K. G. Silvey is Distinguished Professor, Chairman of Biological Sciences, and Director of the Institute for Environmental Studies at North Texas State University, Denton. He received his B.S. from Southern Methodist University in 1927, his M.A. in 1928 and his Ph.D. in Zoology from the University of Michigan in 1932. He is a specialist in eutrophication and in reservoir ecology with reference to the effect of microorganisms on taste and odors.

J. Edward Singley is a Professor in the Department of Environmental Engineering Sciences at the University of Florida. He received his B.S. and M.S. degrees in Chemistry from Georgia Institute of Technology in 1950 and 1952, respectively, and his Ph.D. degree in Water Chemistry from the University of Florida in 1966. He is a specialist in the chemistry of water and wastewater treatment.

Richard L. Woodward is Vice President of Camp Dresser & McKee Inc., consulting engineers in Boston, Massachusetts. He received a B.S. in Civil Engineering from Washington University, St. Louis, Missouri, in 1935, an M.S. in Sanitary Engineering from Harvard University in 1948, and a Ph.D. in Physics from the Ohio State University in 1952. He specializes in water quality problems and water and wastewater treatment.

Panel on Freshwater Aquatic Life and Wildlife

John Cairns, Jr. is University Professor of Zoology and Director of the Center for Environmental Studies, Virginia Polytechnic Institute and State University, Blackburg, Virginia. He received an A.B. from Swarthmore College in 1947, and an M.S. in 1949 and a Ph.D. in 1953 from the University of Pennsylvania. He is a specialist in the effects of stress upon aquatic organisms.

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Panel on Marine Aquatic Life and Wildlife

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Panel on Industial Water Supplies

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FDA (*see* U.S. Department of Health, Education, and Welfare, Food and Drug Administration)
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